Independent Investigation into Water Quality and Local Current Flows at Mullaloo Beach

A distant aerial view of Ocean Reef Marina, Mullaloo Beach, Pinnaroo Point, and Hillarys Marina, facing north

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Acknowledgement of Country

O2 Metocean acknowledges Aboriginal and Torres Strait Islander people as this land's first storytellers and holders of scientific knowledge through their ongoing and continued connection to land, sea and community. We pay our respect to Elders past and present for their custodianship of the land and sea over millennia, which inspires us daily in our collective responsibility to sustain the land and sea country which we live by, work in and dream about.

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Changes from Previous Versions

Changes beyond amendment of typographical and formatting errors, or adjustment of text for clarity are listed below.

Previous Document Revision and Section	Updated in Document Revision and Section	Change
-	-	-

Acronyms and Abbreviations

Acronyms & Abbreviations	Definitions
AHD	Bathymetries are relative to Australian Height Datum
B	Initial buoyancy
BOM	Bureau of Meteorology
BTEXN	Benzene, toluene, ethyl benzene, xylene, naphthalene
Cb	Concentration of a non-reactive constituent in the receiving environment
Cf	Concentration of a non-reactive constituent in the final solution
Chl-a	Chlorophyll-a
Cs	Concentration of a non-reactive constituent in the sample
CTD	Conductivity, temperature, depth
D	Dilution factor
DGV	Default guideline values
DOT	Department of Transport
DWER	Department of Water and Environmental Regulation
ECMWF	European Centre for Medium-Range Weather Forecasts
EPA	Environmental Protection Authority
EQC	Environmental Quality Criteriaa
EQG	Environment Quality Guideline
EQMF	Environmental Management Framework
EQOs	Environmental Quality Objectives
EQS	Environmental Quality Standard
ERA5	ECMWF reanalysis version 5
EVs	Environmental Values
g	Acceleration due to gravity
GA	Geoscience Australia
HD	Hydrodynamic
HEPA	High ecological protection area
IOA	Index of Agreement

Acronyms & Abbreviations	Definitions
LAC	Light attenuation coefficient
LC	Leeuwin Current
LEP	Level of ecological protection
LEPA	Low ecological protection area
l_m	Momentum length scale
LOR	Limit of Reporting
M_o	Initial momentum
MHHW	Mean Higher High Water
MLLW	Mean Lower Low Water
MSE	Mean Square Error
MSL	Mean Sea Level
NHMRC	National Health and Medical Research Council
NWQMS	National Water Quality Management Strategy
NOEC	No Observed Effect Concentration
O2M	O2 Marine (Environment)
O2Me	O2 Metocean
OR	Ocean Reef
ORR	Ocean Reef reference
ORT	Ocean Reef transects
ORSL	Ocean Reef shoreline
Ortho-P	Orthophosphate
OZE	Observed zone of effect
PAH	Polycyclic aromatic hydrocarbons
PEU	Phytoplankton ecology unit
PLOOM	Perth Long Term Ocean Outlet Monitoring
PVD	Progressive Vector Diagram
Q	Volume flux or flow rate

Acronyms & Abbreviations	Definitions
RANS	Reynolds-averaged Navier Stokes
RMSE	Root Mean Squared Error
SSC	Suspended solids concentration
TWW	Treated Wastewater
TN	Total Nitrogen
TP	Total Phosphorus
TPXO	A series of fully-global models of ocean barotropic tides which best fit the Laplace Tidal Equations and assimilated datasets.
TRH	Total recoverable hydrocarbons
TTC	Thermotolerant coliforms
TTM	Total toxicity of the mixture
Vs	Volume of a non-reactive constituent in the sample
Vb	Volume of a non-reactive constituent in the receiving environment
Vf	Volume of a non-reactive constituent in the final solution
w	Mean discharge velocity of the jet
WA Health	Department of Health WA
WASQAP	Western Australian Shellfish Quality Assurance Program
WaterCorp	Water Corporation of Western Australia
WET	Whole effluent toxicity
WRRF	Water Resource Recovery Facility
z	Distance from point of discharge
$\Delta\rho$	Density gradient between the effluent and seawater
ρ	Density of the seawater
2D	Two dimensional
3D	Three dimensional

Executive Summary

A hydrodynamic modelling study was undertaken to assess the influence of the Ocean Reef Marina redevelopment on local currents and the mixing of the Beenyup Wastewater Treatment Plant effluent. The study was prompted by community concerns regarding the potential effects of the Ocean Reef Marina redevelopment. A concurrent community consultation process revealed that the public associates the marina redevelopment with changes in hydrodynamic circulation patterns, which they believe would affect the distribution of treated wastewater discharged through the Beenyup Wastewater Treatment Plant outfall. It is further alleged that these changes could explain the increase in algal growth and perceived decline in marine water quality along the coastline between Burns Beach to Hillarys Boat Harbour. These concerns framed the objective of this hydrodynamic modelling study, which aimed to assess:

- Whether the proximity and layout of the redeveloped breakwater at the Ocean Reef Marina have altered the fate and transport of the Beenyup effluent, relative to pre-development conditions.
- Whether the redeveloped Ocean Reef Marina has decreased the flushing of Mullaloo Beach water, potentially contributing to the recent reduced water quality events, relative to pre-development conditions.

Other plausible causes for the alleged deterioration of water quality along the coastline have been proposed (e.g. structural integrity of the pipeline and potential leakage of effluent closer to shore, overall warmer water temperatures, sediment plumes originated at the marina construction site, other nutrient sources, etc.), but the connection between these causes and the water quality issues reported were not investigated in this study.

To support the assessment, the Department of Water and Environmental Regulation (DWER) provided water quality datasets for critical review to characterise changes in the composition of the treated wastewater (TWW) and assess the influence of the discharge on marine water quality. Minimal detectable changes in the discharge volume and constituent concentrations discharged were identified, which could be considered causes for concern. However, higher organic particulates were measured in the TWW in 2024.

Over the past three years, natural background concentrations of chlorophyll-a in the region have increased, raising concerns about potential nutrient loading. This trend is evident despite the lack of significant changes in the characteristics of the treated wastewater (TWW) during the same period. The TWW is discharged with elevated nutrients, however, monitoring of the discharge has demonstrated that elevated concentrations are gradually reduced with distance from the outfall in the direction of the current and are barely detectable above background levels at 1,500 m. The TWW signal rarely tracks towards Mullaloo Beach in summer, therefore nutrient inputs to the beach south of the redeveloped marina from the Beenyup discharge are likely to be minimal. In receiving waters, toxicants were low and physico-chemical data represented natural marine conditions. Elevated *Enterococci* concentrations recorded at shoreline sites on 23 January 2024 indicate a potential local source of faecal contamination, but its connection to the Beenyup effluent could not be confirmed based on the available information. Other anthropogenic sources such as sanitary waste systems aboard boats or poorly maintained nearby pump-out stations should not be discarded.

Toxic phytoplankton listed under the Western Australian Shellfish Quality Assurance Program (WASQAP) commonly occur within samples reviewed, with *Pseudo nitzschia* spp. exceeding guideline alert levels in 2024. The 'WA Health watch list' Cyanobacteria *Trichodesmium* was present in samples from Mullaloo Beach on 17 January,

and 24 and 26 of April 2024, likely the cause of the brown surface scum identified in public complaints. *Trichodesmium* is a fixer of nitrogen, able to convert nitrogen into a form bioavailable for phytoplankton and is not necessarily related to nutrient loading. Therefore, no definitive connection could describe the influence of the discharge on recent marine water quality issues, and data available are limited to evaluate further.

Hydrodynamic modelling was conducted using the Mike software suite developed by the DHI Group, which solves the unsteady coupled Reynolds-averaged, hydrostatic, Navier-Stokes equations and the scalar equation for salinity, temperature, turbulent kinetic energy and tracers. A literature review revealed that wind-driven circulation dominates at the study site. Hence, the analysis of the optimal model configuration focused on identifying the numerical domain(s), wind field(s), and wind drag coefficient that would produce modelled currents which best replicated the observed current field at the study site. Model accuracy was validated using a set of skill scores including model bias, mean square error (MSE), root mean squared error (RMSE), and the Wilmott (1981) Index of Agreement (IOA), along with a qualitative evaluation of Progressive Vector Diagrams (PVD). Optimal model results were achieved using a nested approach, in which a three-dimensional (3D) 'local' model was embedded within a larger two-dimensional (2D) 'regional' model domain. The regional model was forced with a spatially and temporally varying wind field sourced from the European Centre for Medium-Range Weather Forecasts (ECMWF) reanalysis v5 (ERA5), and astronomical tides reconstructed from TPXO Global Tidal Model outputs at its single open boundary. The local model, on the other hand, was forced with wind data measured at the Bureau of Meteorology (BOM) Ocean Reef weather station and water level fluctuations extracted from the regional model. Two comparable meshes were prepared for the local model: one representing the pre-development and the other post-redevelopment breakwaters of the Ocean Reef Marina. The bathymetry was compiled from multiple products, prioritising Department of Transport's (DOT) high-resolution Lidar and multibeam observations around Ocean Reef. The length scale of the numerical grid elements ranged from 9,000 m at the offshore open boundary of the regional domain, to 12 m around the Beenyup Wastewater Treatment Plant outfall in the local domain. The effluent was assigned temperature and salinity values consistent with freshwater density of approximately 1,000 kg/m^3 (Pattiaratchi, 1991), while measured values were used to represent the surrounding marine environment. A value of 0.1 m^2/s was chosen for the horizontal eddy diffusivity of salinity, temperature, and numerical tracer, based on previous studies in the area (Pattiaratchi, 1991, Apasa, 2011).

The effect that the post-development Ocean Reef Marina has on local hydrodynamics was investigated by computing the difference in modelled current speeds relative to pre-development conditions during representative summer and winter wind regimes. During summer mornings, wind-driven circulation has changed minimally post-development of the marina, as the marina causes limited disruption to the fetch of the predominant easterly winds. Later in the day, as the wind shifts direction, the interaction between the new marina layout and the south-westerly sea breeze results in a net reduction in depth averaged current speeds of 0.05-0.10 m/s leeward of the marina (compared to pre-development conditions), affecting a region extending approximately 1,500 m to the north. A relatively small area extending approximately 500 m to the south of the marina and 250 m offshore the beach is also affected, with currents being weaker compared to pre-development conditions. Alongshore, northerly flows must negotiate the new marina resulting in a modest increase in current velocities west of the redevelopment, extending toward the Beenyup diffuser site. Current flows around the redeveloped marina during a typical calm day in winter shares resemblance with the pattern experienced during a summer day: depth averaged currents are lower to the north and south of the marina relative to pre-development conditions, while increased currents are evident to the west of the marina. The disruption of a southerly flow results in a slightly larger area of depth average current reduction south of the marina (1,200 m), compared to the reduction north of the marina (1,100 m). Length scales expressed in this section were derived for a 10% reduction contours and are indicative only since they are based on limited but representative seasonal behaviour. The results demonstrate, nevertheless, that weaker flows are expected both north and south of the marina relative to pre-development conditions, aligning with anecdotal reports of areas experiencing poor water quality.

A qualitative assessment of the area of influence of the Beenyup effluent was conducted by examining the spatial extent of an idealised effluent released through the Beenyup diffusers under various idealised wind forcing conditions, for pre- and post-development model configurations. The idealised cases modelled showed no evidence of interaction of the Beenyup effluent with the coastline north and south of Ocean Reef Marina at up to 500-fold dilutions, subject to feasible, persistent, summer wind conditions of up to 12 hr duration. The numerical results of idealised conditions suggest that atypical summer metocean circumstances would be required for treated wastewater discharged through the Beenyup outfall to reach the coastline at concentrations below 500-fold dilution. For example, wind events persisting for more than 24 hours (occurs approximately 1% of the time during summer, or approximately once over this period) from the north-west direction (experienced less than 3% of the time during summer) would be required. Most importantly, the results revealed that under the wind directions and intensities considered, there were negligible differences in the extent and location of the Beenyup effluent dilution contours when post-redevelopment results were compared to pre-redevelopment conditions. No evidence has been found to support the claim that the marina redevelopment has substantially altered the distribution of the dispersed Beenyup effluent near the coastline. The perceived changes to water quality may also be influenced by other natural and anthropogenic factors, which may include changes in local winds, current speeds, overall warmer water temperatures, local groundwater seepage and nonpoint source pollution from the marina.

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1. Introduction

1.1. Background

The City of Joondalup is redeveloping the old Ocean Reef Boat Harbour into a world class marina featuring recreational, boating, residential and tourist precincts. Construction of the multi-million-dollar Ocean Reef Marina redevelopment commenced in August 2020. The southern breakwater reached its final length of 1180 m in December 2022, and the northern breakwater was completed in May 2022. The old breakwater was removed in March 2022. The final layout of the marina relative to the pre-redevelopment layout is shown in Figure 1.

Figure 1: Nearmap satellite image of the Ocean Reef Marina in October 2024 relative to the (manually traced) pre-redevelopment layout from March 2020, shown with a yellow dashed line.

The *Trichodesmium* algal bloom event at Mullaloo Beach in January 2024 (Figure 2), along with alleged recurring water quality issues at both Mullaloo and Ocean Reef beaches before and after this event (e.g. Figure 3), has raised public concerns. Some community members have suggested that the redevelopment of the Ocean Reef Marina may have altered hydrodynamic circulation patterns in this region. These changes are alleged to have caused a redistribution of nutrients and other constituents discharged by the Beenyup Wastewater Treatment Plant outfall (hereafter referred to as 'Beenyup effluent'), potentially explaining the observed incidents—including a reported increase in algal presence and a decline in marine water quality (O2Me 2024). The location of the discharge pipelines and approved Low Ecological Protection Area (LEPA) are shown in Figure 4.

Figure 2: *Trichodesmium* bloom in Mullaloo beach in January 2024 (K Allen 2024, incorporated into O2Me 2024)

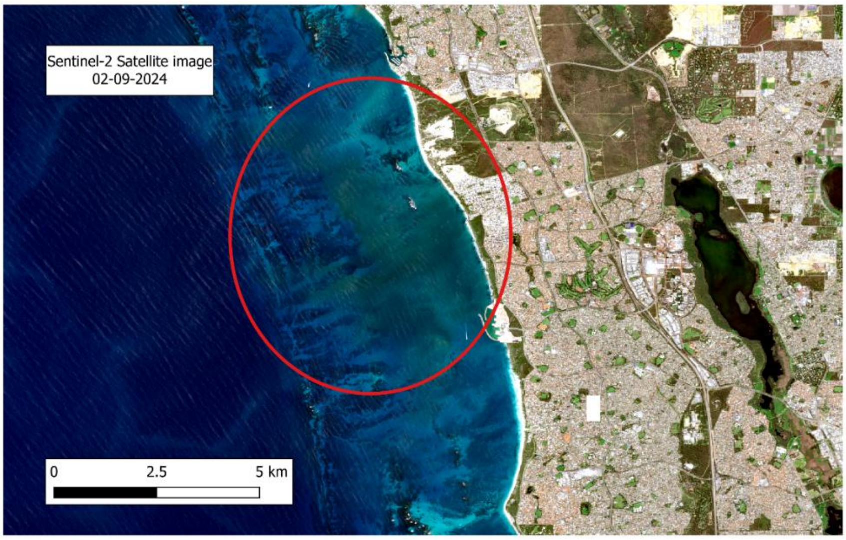

Figure 3: Sentinel-2 satellite image suggesting discoloured water mass north of Ocean Reef Marina, allegedly caused by the Beenyup outfall (K. Allen 2024, incorporated into O2Me 2024)

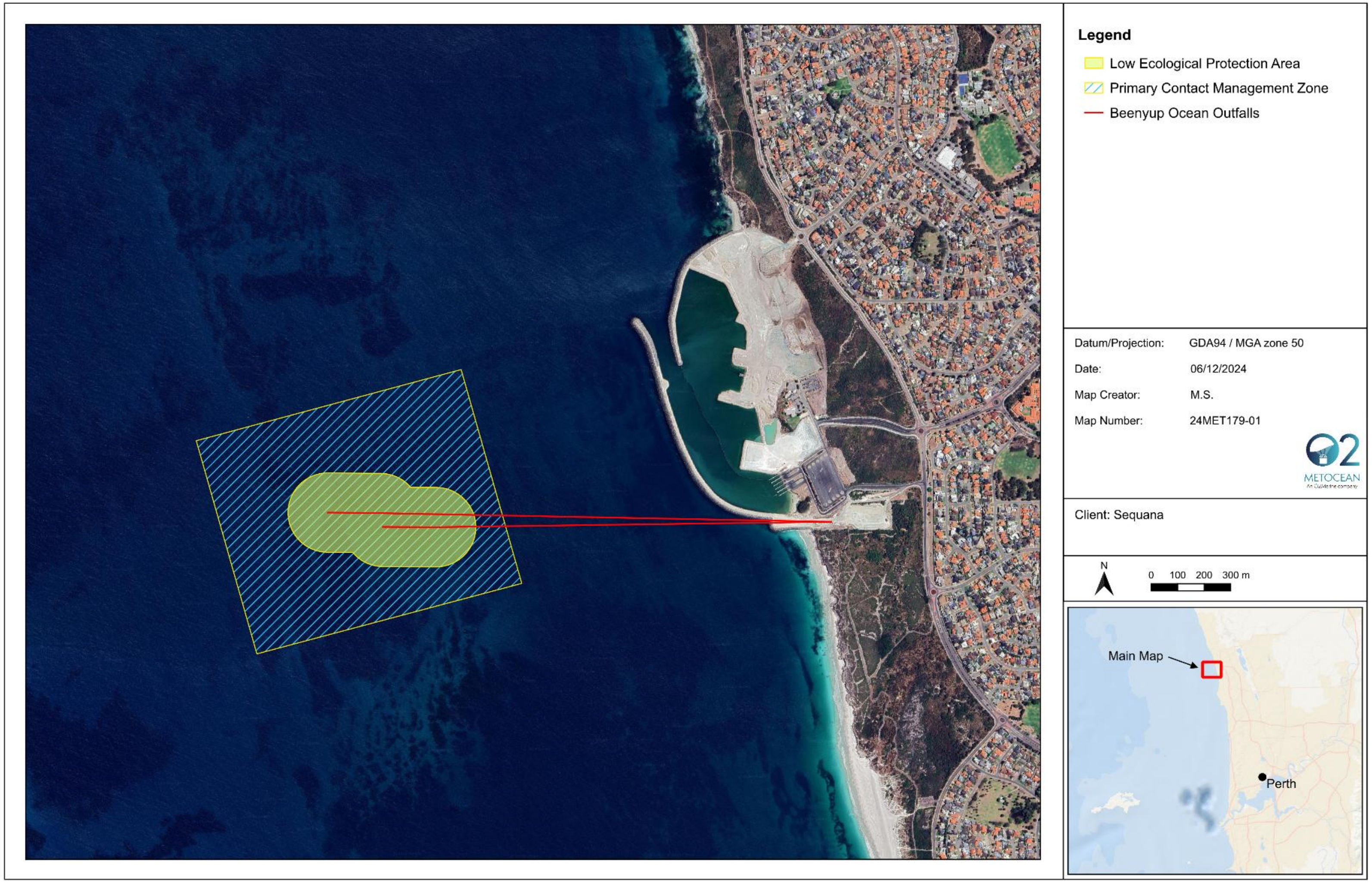

Figure 4: Beenyup Wastewater Treatment Plant discharge pipelines, EPA approved Low Ecological Protection Area and Primary Contact Management Zone.

To investigate if the allegations made by the public are substantiated, the Department of Water and Environmental Regulation (DWER) engaged O2 Metocean (O2Me) to conduct a community consultation, analyse marine water quality datasets, and undertake a hydrodynamic modelling exercise of the Mullaloo and Ocean Reef local environment. A summary of the common themes identified through the consultation process is provided in the accompanying O2Me (2024) report. This report (this document) focusses on the hydrodynamic model developed for the study, including a chapter dedicated to the analysis of marine water quality datasets.

1.2. Framing the Hydrodynamic Modelling Scope through Community Consultation

O2 Marine (O2M), an affiliated company to O2Me, facilitated a community consultation process to ensure the voices of the general community and industry stakeholders were heard and documented. The community concerns raised through the community consultation process were grouped into thematic issues as summarised in O2Me (2024), to be further considered during the analysis of readily available water quality data and the concurrent hydrodynamic modelling study. “Concern for the Environment” was the highest-ranking theme of concern among the community, which relates to secondary themes of concern about “Public Health” and “Governance/Communication”. O2Me (2024) highlighted the importance of framing the marine environment issues at Mullaloo Beach from a wider perspective when responding to community concerns, so the community can better understand the context of the issue. A primary example given by O2Me (2024) was the need to provide wider contextual information on topics such as climatic conditions to assist community level understanding of the issues they experience around Mullaloo and Ocean Beach.

As only preliminary findings of the community consultation process were available when this hydrodynamic study commenced, the outputs from the community consultation meeting held on 29 August 2024, summarised in Table 1, were considered in the development of the hydrodynamic model.

Table 1: Concerns raised during the community consultation workshop and perceived associated causes identified through group discussion on 29 August 2024 [*replicated from Table 3 in O2Me 2024*]

Theme	Concerns raised	Perceived Causes

Public health	• Illness (including respiratory health, long-term health and mental health)
	• Pet health
	• Toxic seafood
		• Increase illness when a wastewater 'flush' takes place
		  ◦ No consistent days for when a 'flush' occurs
		• Outfall pipe conditions
		  ◦ Holes along the outfall pipe (seen from drone footage)
		  ◦ The proximity of the end of the pipe to the coastline has been reduced with the building of the Ocean Reef marina breakwater
		  ◦ Structural integrity
		  ◦ Pipe hit during construction of marina
		  ◦ Age of pipe
		  ◦ Screen not maintained
		• Warmer days
		• When current comes from the direction of Ocean Reef
		• Evidence noticed more on still days highlights the problem
		• Increase population in wastewater catchment

Environment	• Water quality/odour
	• Tipping point (now or in the future)
	• Deceased fauna (including fish, seabirds, leafy sea dragon)
	• Ecosystem health
	  ◦ BCH/Seagrass/reduction in wrack
	  ◦ Sediment transport/shoreline changes/changes to the offshore sand banks
	  ◦ Currents (onshore/offshore)
	  ◦ Microbiocidal health
	  ◦ Anaerobic conditions
	• Waste (including cotton buds, rubbish, microplastics, PFAS)
		• Outfall
		• Development of Ocean Reef Marina
		• Pollution
		• Climate change
		• Changes in pH? Deteriorating the limestone materials used for the Ocean Reef marina breakwater
		• Sediment plume from the marina
		• Increased turbidity

Community reputation	• Stigma/name-calling • Loss of access/ beach closures • Economic (including retail/housing/club memberships) • Indigenous engagement • Social • Recreational (fishing, etc…)	• Outfall discharge ◦ Increased nutrients? • Water currents (wind) • Seasons • Water temperature • Water stagnation • Greater discharges • Contaminated seafood/ suitability of consumption • Reduced visibility

Theme	Concerns raised	Perceived Causes
	• Education/swimming classes cancelled • Hospitalisations	• Unusual swell • Changes in benthic habitats/loss of habitats • Lack of reporting and communication of reporting • Lack of appropriate and timely action
Reporting	• Sampling protocols • Compliance/triggers • Test results • Discharge quality/treatment • Analytes • Baseline (pre/post)	• Sediment plume monitoring • Lack of frequency and consistency of onshore and offshore sampling • Limited analyte and microbial testing (not testing for PFAS, Protozoa, Giardia) • Cotton bud stem reporting

Governance/ Communication	• Engagement/Communication
	• Transparency
	• Work Health and Safety (WHS)
	• Accountability
	• Outfall
	  ◦ Suitability of outfall
	  ◦ Condition of outfall
	• Licencing/approvals (outfall)
	• Agency roles/responsibility
	• Ocean Reef Marine documents
	• CHRMAP
	• Department of Transport (DoT) coastal development
	• Reputation/risk
		• Lack of clarity of
		  ◦ Roles and responsibilities
		  ◦ Regulation
		  ◦ Policy and procedures for testing and reporting of results
		  ◦ Duty of care (for the environment)
		  ◦ Public advocacy
		• Inadequate licence and conditions of licence (operating licence)
		• Inadequate management actions when trigger exceedance occurs
		• Inadequate reactive management processes
		• Lack of baseline data
		• Transparency with the community

A heatmap of the complaints made to DWER via Pollution Watch between August 29, 2024 and September 30, 2024, are summarised in Figure 5 (refer to Appendix A).

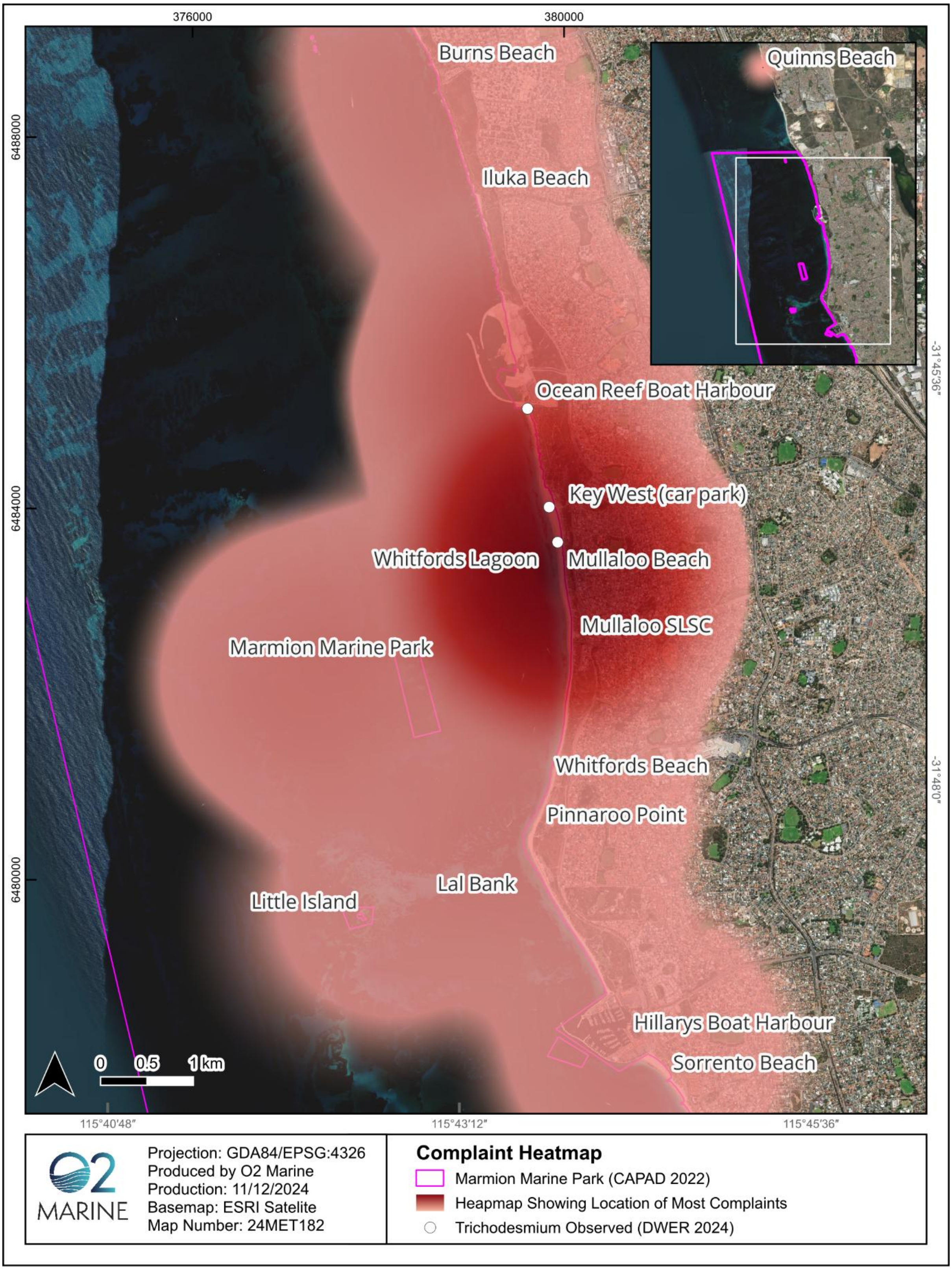

Figure 5: Complaint heatmap. Developed from public submissions received during the period August 29 to September 30, 2024.

The concerns raised by the community and the perceived associated causes presented in Table 1, prompted DWER need to know if:

1. The proximity and layout of the redeveloped breakwater at the Ocean Reef Marina have changed the fate and transport of the Beenyup effluent, relative to pre-development conditions.
2. The redeveloped Ocean Reef Marina altered ocean currents which previously (i.e. pre-redevelopment) could have assisted in flushing Mullaloo Beach water, potentially leading to the recent reduced water quality events.
3. The composition of the Beenyup effluent (nutrients or other constituents) has changed (over the past few years).
4. Other factors such as increased ocean water temperatures, the effects of a growing population in the Beenyup Wastewater Treatment Plant catchment area, or other nutrient sources, could have led to the observed algal bloom events.
5. Construction activities associated with the Ocean Reef Marina redevelopment are detrimental to the environment (e.g. sediment plumes from the marina).

In this report, O2Me integrated numerical modelling results with other supporting evidence to address items 1 and 2, above. A desktop review of existing data on the Beenyup Wastewater Treatment Plant has been undertaken to address item 3. Through discussions with DWER, O2Me acknowledges that addressing Points 4 and 5 was beyond this scope of work.

1.3. Objective of this Report

To investigate if the redevelopment of the Ocean Reef Marina could have substantially altered localised ocean currents and the distribution of constituents discharged by the Beenyup effluent, compared to pre-redevelopment conditions.

1.4. Scope of Work

The scope of work includes:

1. Conduct a literature review and data analysis to identify the oceanographic mechanisms dominating mixing and dispersion around the Ocean Reef Marina to incorporate into the numerical model.
2. Undertake review of data provided by the Water Corporation of Western Australia (WaterCorp) relating to the local marine water quality and outfall discharge quality to characterise changes in the composition of the Beenyup effluent and assess the influence of the Beenyup ocean outfall on marine water quality.
3. Conduct an independent assessment of near-field mixing conditions for the Beenyup Wastewater Treatment Plant outfall discharge, to determine the optimal cell size for the far-field numerical model.
4. Perform a sensitivity analysis to evaluate the model's response to different characterizations of the key forcing mechanism, identifying the optimal setup conditions for the model.

5. Run the far-field numerical model for a range of scenarios targeting the objective of this report.
6. Assess if the model results support or challenge the public's perception that the redevelopment of the Ocean Reef Marina is partly responsible for the recent observations of poor water quality south of the marina.

1.5. Approach

A three-dimensional 3D hydrodynamic numerical transport model was developed for pre- and post-Ocean Reef Marina redevelopment layouts. The validated model was forced with the predominant summer forcing mechanisms responsible for advection and dispersion in the region. The Beenyup effluent was represented in the model using a constant *numerical signal* ([1]), with its concentration tracked over time and distance to evaluate dilution as a function of both. An assessment of the potential for the new marina layout to affect local hydrodynamics at Mullaloo and Ocean Reef beaches as well as to modify the dispersion characteristics of the Beenyup outfall was made. In addition, a desktop review of data provided by WaterCorp and other local water quality data publicly available was performed to characterise changes in the discharge wastewater stream and support modelling outputs. Findings were discussed in the context of public concerns captured during the community consultation process.

1.6. Exclusions and Limitations

Excluded from the scope of work, and therefore this report, are the following:

- Fate and transport studies of sediment plumes arising from the construction of the Ocean Reef Marina or resulting from its revised layout.
- Diffuser design or optimisation.
- Engineering assessment of pipeline integrity and/or dispersion modelling of unplanned effluent leaks from alleged holes in the outfall pipe.
- Dye dispersion, drift drogue, or particle tracking studies to derive local horizontal eddy diffusivity and advection characteristics.
- Assessment of impacts to marine water quality, benthic communities, general population, or other, resulting from the revised marina layout.
- Independent collection of samples for additional water quality data.

1.7. Definitions, Conventions, and Datums

'Mixing' is the physical process responsible for the scattering of particles or a cloud of diluted contaminants by the combined effect of shear and transverse turbulent diffusion, a process which causes one parcel of water to be mingled with or diluted by another.

'Mixing zone' is the area around an ocean outfall where a certain environmental quality criterion is exceeded.

'Near-field mixing' is a fluid dynamics concept used to describe the mixing that an ocean outfall experiences due to the characteristics of the discharge, often based on the dominant mixing processes

[1] refer to Section 1.7 for a definition of 'numerical signal'.

of a jet or plume. A '**jet**' is driven by the momentum of the discharge, whereas a '**plume**' is driven by the potential energy of the discharge providing the fluid with a positive or negative buoyancy relative to its surroundings (Fischer et al. 1979).

The '**end of the near-field**' is the point where mixing ceases to be dominated by differential momentum or buoyancy between the discharge and receiving environment and mixing by background turbulence begins to take over. There is no objective definition for the end of the near-field, and it cannot be resolved by a near-field only model. Numerical models that resolve jet and plume mixing dynamics are referred to as '**near-field models**'.

'**Far-field mixing**' is also a fluid dynamics concept that describes the mixing and transport of the discharge, away from the near-field, primarily due to natural processes (ambient hydrodynamics). Hydrodynamic models that solve the unsteady Reynolds-averaged Navier Stokes (RANS) equations of mass and momentum, incorporating non-hydrostatic and baroclinic pressure gradients, Coriolis effects, etc. are one type of '**far-field models**'.

A '**Numerical Signal**' in the context of numerical modelling refers to a simulated, non-reactive, and conservative quantity used to track the movement and dispersion of substances within a modelled system. Governed by the advection-dispersion equation, this signal behaves passively, meaning it does not undergo chemical reactions, decay, or transformation, and has no internal sources or sinks during its modelling.

'**Dilution factor**' (D) is the factor by which the volume of a sample (e.g. outfall discharge) is diluted with ocean water, and is derived from the simple conservation of mass equation:

$$C_f V_f = C_s V_s + C_b V_b$$

Where *Cs*, *Cb*, and *Cf* are the concentrations of a non-reactive constituent in the sample, receiving environment, and final solution, respectively, and *Vs*, *Vb*, and *Vf* are their respective volumes.

In hydrodynamic numerical fate and transport studies, it is customary to "release" a *numerical signal* with an initial concentration *Cs*=1 and nominal volume *Vs*=1 into the surrounding environment initialised with a concentration of *Cb*=0, such that:

$$C_f (V_b + V_s) = C_s V_s$$

$$V_b = V_s \left(\frac{C_s}{C_f} - 1 \right)$$

The dilution factor, *D*, is then reported as "$D = 1{:}V_b$" and interpreted as "1 part of sample to *Vb* parts of dilutant".

In practical terms, the concentration of the numerical signal at every numerical grid cell and timestep, *Cf*, is tracked allowing for the calculation of dilution factors. By way of example, if the target dilution factor is 1:100, also referred to as 'one-hundred-fold dilution', the concentration of the numerical signal in any numerical cell that satisfies the criterion $C_f = C_s V_s/(V_b + V_s) = 1 \cdot 1/(100 + 1) = 0.009901$ meets the specified dilution requirement.

Directional Acronym Conventions: When describing the directionality of metocean parameters (such as wind, waves and currents), acronyms for direction are used in the report. For example, a wind direction may be described as SSE instead of a south-southeast. The exceptions to this convention include:

- Usage in preceding Sections, prior to introduction of this convention.
- When describing direction within a header.
- When describing a proper noun (such as ‘Southwest Regnard Island’ or ‘Southwest Trade Winds’).
- When describing the direction that is not related to measured data or metocean parameters (such as describing a location for example ‘southern Australia’).

Datums: Water depths and water levels are relative to mean sea level (MSL). Bathymetries are relative to Australian Height Datum (AHD). Maps are created in WGS 84 - Pseudo-Mercator (EPSG:3857) and GDA 2020 MGA Zone 50 (EPSG:7850) coordinate reference system unless otherwise stated and are not to be used for navigational purposes. Positional accuracy should be considered as approximate.

2. Oceanographic Context

2.1. Geomorphology

The geology and geomorphology of the greater Perth Metropolitan coastline have been extensively detailed by Searle & Semeniuk (1985) and Richardson, et al. (2005). Searle & Semeniuk (1985) classified the coast into several sectors, with the study area falling at the southern end of the Whitfords to Lancelin sector. The coast in this sector is characterized by various features including rocky coasts, pocket beaches situated along with straight coasts and sandy beaches. Onshore geomorphology is featured by mainly four areas of limestone ridges which are nearly parallel to shoreline. One forms the architecture of rocky shores and extend up to 150m seawards. The other three limestone ridges are named Staggie Reef Ridge, Marmion Reef Ridge and Spearwood Ridge which are located 6, 4, 2 km offshore respectively. In the local study area, there are offshore reef platforms north of Hillary's Boat Harbour, an isolated accretionary cusp at Pinnaroo Point, and sandy beaches and dune systems extending to Mullaloo. From Mullaloo to Burns Beach limestone cliffs and bayed beaches are present, with a predominantly sandy beach and nearshore reef platforms north of the Burns Beach groyne (M P Rogers & Associates, 2016).

The study area is also part of Rottnest region (Richardson et al., 2005). The continental shelf within this region covers an area of 43,500 km^2 and includes the Rottnest Shelf and Dirk Hartog Shelf, extending to the Perth Abyssal Plain. The Dirk Hartog Shelf, approximately 70 km wide, stretches from North West Cape to Shoal Point (Harris et al, 2005). The Rottnest Shelf spans from Geraldton to Cape Leeuwin and is 45-100 km wide. It is a narrow, flat-topped shelf that slopes steeply away from the coast, supporting tropical reefs at the Houtman Abrolhos Islands and cool-water carbonate production on the shelf and upper slope. The surface sediments are arranged parallel to the shelf, and the continental slope is incised by numerous canyons, with the largest being the Perth Canyon offshore of Rottnest Island, serving as a significant biogeographical boundary. This canyon intersects the shelf and channels detritus and sediment into deeper waters.

The nearshore topography includes eroded limestone reefs and pinnacles that rise 10-20 m above the seabed, extending up to 5 m from the sea surface. These formations provide coastal protection from wave energy and contribute to the development of cuspate forelands, embayment, and inlet beaches (Sanderson et al., 2000). The inner shelf plain is gently sloping and smooth, with water depths less than 48 m.

Near the Beenyup discharge and Ocean Reef Marina up to ~1.5 km offshore, the seabed is characterised by a relatively uniform seabed with some alternated rocky and sandy bottom. Small rocky crops and ridges raise up to 1 m high in this region and may play a small role in local hydrodynamics. More prominent seabed features can be found south of the Beenyup diffuser, where a ~0.5 km^2 patch of elevated rocky features rising to 4 m from the average seabed is present. Small canyons crossing this patch are up to ~2 m deep. Further offshore, ~2 km away from the study site, is the rocky Marmion Ridge and beyond it, the seabed gently slopes down to ~-20 m (AHD) westward.

2.2. Wind

Perth coast climate is defined by hot and dry summers along with wet and mild winters. Main weather events usually happen during either the summer or winter season, with spring and autumn being transitional. Annual movements of the anticyclonic belt control the wind climate around the Ocean Reef Marina. Summer (December to March) is usually characterised by predominant north-easterly winds in the morning and south-westerly sea breeze in the afternoon. Winter (May to October) usually presents significant rainfalls generated by cold fronts forming from the low-pressure systems approaching the coast form the west/south-west.

Bureau of Meteorology (BOM) Ocean Reef (BOM ID: 009214) and Rottnest Island (BOM ID: 009193) weather stations were considered relevant to understand and model the local wind conditions. The last 7 years (January 2018 to October 2024) were considered in this modelling study, such that the datasets overlapped with other oceanographic datasets available.

Table 2 and Table 3 present the monthly, summer (December, January and February), and winter (June, July and August) median wind intensities for each cardinal direction, as measured at Ocean Reef and Rottnest Island weather stations, respectively. Due to the importance of wind directionality in this assessment, the joint frequency distribution of wind intensity by direction for the summer months (December, January and February) is shown in Figure 6. Table 4 summarises the persistence exceedance of wind intensity for a representative summer period (December 2022, January and February 2023). Monthly wind roses are shown in Figure 7 and Figure 8.

The well-known seasonality to the wind pattern around Perth is evident in the dataset. Summer winds typically exhibit higher median intensities from the S and S-W than any other cardinal direction. In addition, summer winds predominantly (66.3%) arrive at the study site from the southeast to west (135°-270°), while winds from the west to north (270°-360°) constitute only 2.8% of the measured directions (Figure 6). The remaining 30.9% of the summer winds originate inland (0°-135°) and tend to 'push' surface coastal waters offshore. Persistence exceedance analysis of the wind record revealed a characteristic daily wind cycle during summer (Table 4). Sustained winds exceeding 4 m/s regularly persisted for up to 6-hours (50.02%), often up to 12-hours (27.70%), occasionally up to 24-hours (8.86%), and rarely 36-hours (3.02%). Sustained winds exceeding 6 m/s, an intensity adopted in this description to represent the median westerly (8.02 m/s) and south-westerly (7.19 m/s) summer median values, were rare up to 12-hours (7.19%), extremely rare up to 24-hr (~1%) and practically inexistent over a 36-hr period (0.05%).

Winter is characterized by longer periods of low wind intensities and mild easterlies, with a significantly higher likelihood of strong westerly storms (Figure 7 and Figure 8). Though wind patterns are similar at both locations, winds at Rottnest Island are stronger than at Ocean Reef.

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Table 2: Median monthly and seasonal wind speed (m/s) at Ocean Reef BOM station (January 2018 to October 2024). Summer= December, January and February. Winter = June, July and August.

Ocean Reef	N	N-E	E	S-E	S	S-W	W	N-W
January	3.11	5.69	6.69	4.61	8.19	7.69	5.11	3.61
February	3.61	5.69	6.19	4.11	7.69	6.69	4.61	4.11
March	3.61	5.69	5.69	4.11	7.19	6.19	5.11	5.69
April	4.11	4.61	4.61	3.61	6.69	6.19	6.69	6.19
May	4.61	5.69	4.11	3.61	5.69	7.19	8.19	6.69
June	5.11	5.11	4.11	3.61	5.11	7.69	9.31	8.19
July	5.69	5.11	4.11	3.61	5.11	8.69	9.81	8.69
August	5.69	5.11	4.11	3.61	6.19	8.69	9.31	8.69
September	4.61	4.61	4.61	3.61	6.69	6.69	7.19	6.69
October	4.61	4.61	4.61	4.11	7.69	6.69	7.19	7.19
November	4.61	4.11	6.19	4.11	8.19	7.19	6.19	6.69
December	4.11	6.19	6.69	4.11	8.19	7.19	6.69	5.11
Summer	**3.61**	**5.86**	**6.52**	**4.28**	**8.02**	**7.19**	**5.47**	**4.28**
Winter	**5.50**	**5.11**	**4.11**	**3.61**	**5.47**	**8.36**	**9.48**	**8.52**

Table 3: As per Table 2 for Rottnest Island BOM station.

Rottnest Island	N	N-E	E	S-E	S	S-W	W	N-W
January	5.11	6.19	6.69	6.69	9.31	9.31	5.69	4.61
February	5.11	5.69	6.19	6.19	8.69	7.69	4.61	4.61
March	4.61	5.69	6.19	5.69	8.19	7.19	5.11	5.69
April	5.69	5.11	5.11	5.11	7.19	6.19	5.69	6.19
May	6.69	6.19	5.11	5.11	6.69	7.69	7.19	7.69
June	7.69	5.69	4.11	4.61	6.69	8.19	7.19	7.69
July	8.69	5.11	4.11	4.11	6.19	8.19	8.19	9.31
August	7.69	5.11	4.11	4.61	6.69	8.19	8.19	8.19
September	6.69	5.11	5.11	5.69	7.19	7.19	6.19	6.69
October	7.19	5.69	5.69	6.19	8.69	7.19	6.19	6.69
November	5.69	5.11	6.19	6.19	8.69	7.69	5.69	6.19
December	5.69	6.19	6.69	6.19	8.69	8.19	6.19	6.19
Summer	**5.30**	**6.02**	**6.52**	**6.36**	**8.90**	**8.40**	**5.50**	**5.14**
Winter	**8.02**	**5.30**	**4.11**	**4.44**	**6.52**	**8.19**	**7.86**	**8.40**

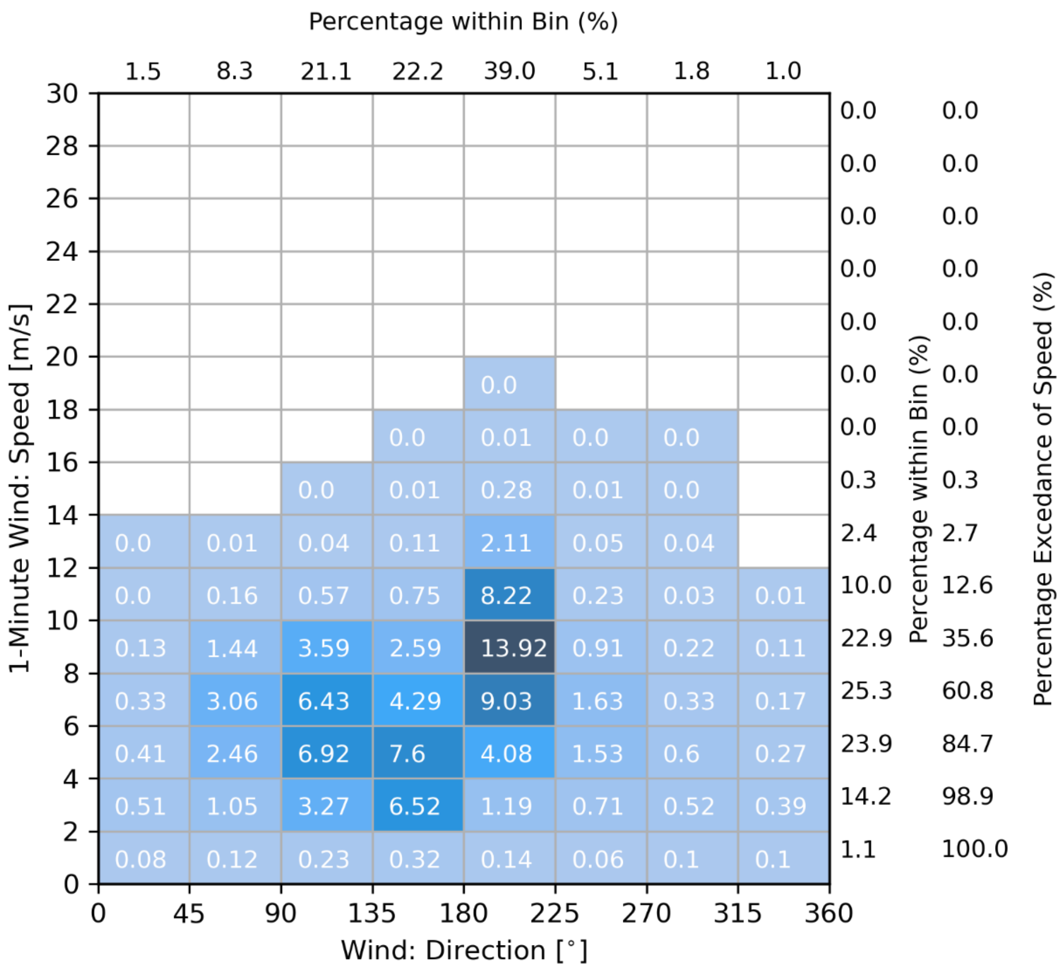


Figure 6: Joint frequency distribution of wind intensity by direction at Ocean Reef Station during the summer months (December, January, February) for the period 2018-2024.

Table 4: Persistence exceedance, as percentage of sampling window, of omnidirectional wind intensity at Ocean Reef Station during the summer months (December, January, February) for the period 2022-2023.

	> 1 m/s	> 2 m/s	> 4 m/s	> 6 m/s	> 8 m/s	> 10 m/s	> 12 m/s	> 14 m/s	> 16 m/s
1-hour	96.09	96.09	77.63	51.69	26.66	9.43	1.75	0.12	0.00
2-hours	94.43	94.43	71.18	43.63	21.56	7.65	1.20	0.06	0.00
4-hours	91.45	91.45	59.75	32.06	14.78	4.95	0.74	0.00	0.00
6-hours	88.93	88.93	50.02	23.44	9.71	2.94	0.43	0.00	0.00
12-hours	81.96	81.96	27.70	7.19	1.38	0.28	0.00	0.00	0.00
24-hours	70.16	70.16	8.86	0.87	0.00	0.00	0.00	0.00	0.00
36-hours	62.98	62.98	3.02	0.05	0.00	0.00	0.00	0.00	0.00

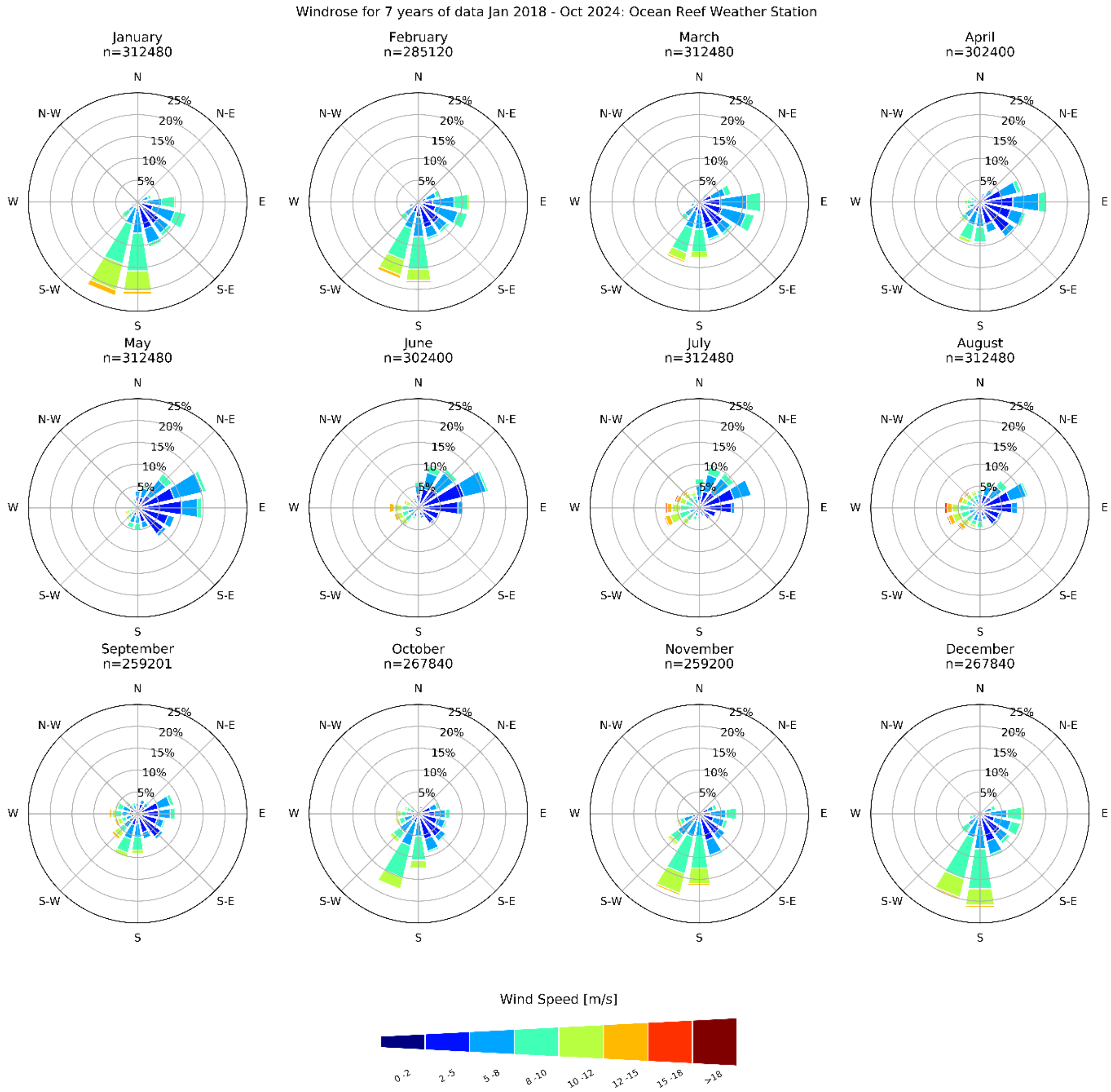


Figure 7: Monthly wind roses for Ocean Reef BOM station from January 2018 to October 2024.

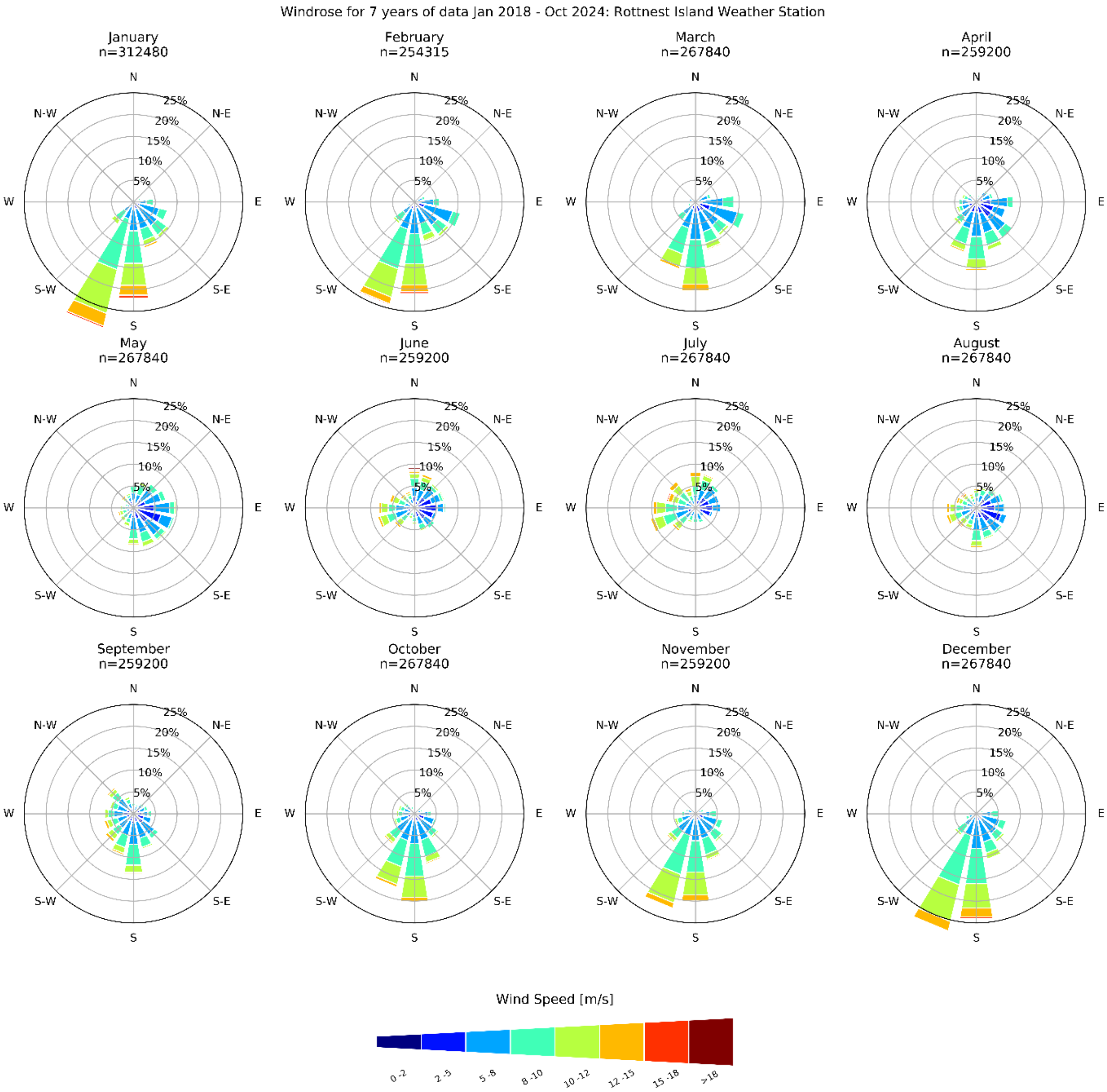


Figure 8: Monthly wind roses for Rottnest Island BOM station from January 2018 to October 2024.

2.3. Water Level Fluctuations

A thorough review and description of the water level fluctuations that may be observed in the vicinity of the Ocean Reef Marina and Beenyup effluent sites can be found in Pattiaratchi (1991). Here, a summary of Pattiaratchi (1991) findings which influence local current circulation is presented.

The Leeuwin Current (LC) drives annual and inter-annual variations in mean sea level, with an amplitude of up to 0.20 m. However, its impact on flushing and water stagnation relative to algal blooms which may develop over timescales of the order of days to weeks is negligible, so no further discussion will be provided ([2]). Mean sea level is also influenced by low pressure systems and the resulting storm surges, occasionally resulting in up to 0.9 m variations.

At the study site, water level oscillations ranging from periods of seconds to days are caused by:

- Continental shelf waves (periods of days): Also referred to as coastally trapped waves, continental shelf waves travel southward over long distances along the continental shelf with amplitudes reaching 0.25 m.
- Tides (order of hours): The typical small tidal range variation of 0.5-0.6 m is explained by the presence of an amphidromic point off the southwest coast of Western Australia. Tides are mainly diurnal, primarily driven by the diurnal tidal constituents O_1 (0.16 m amplitude) and K_1 (0.12 m amplitude). In contrast, the semidiurnal constituents M_2 and S_2 have amplitudes of ~0.05 m and contribute little to the tidal fluctuation at the site. The spring tidal range (MLLW to MHHW) varies from 0.4 to 0.7 m. Pattiaratchi (1991) reports a low alongshore tidal propagation, using the 20 minute difference in the propagation of the diurnal tidal wave between Two Rocks and Fremantle as an example.
- Infra-gravity waves (order of minutes): These low period surface waves have typical periods of 30 seconds to several minutes but play a negligible role in the problem investigated in this study and, therefore, no further discussion will be provided.
- Surface gravity waves (order of seconds): Swells originating from distant storms over the Indian and Southern Oceans reach the study site with periods of 10 to 15 seconds. Seas caused by locally generated winds have a period of 4 to 7 seconds. Wave energy reaching the beach zone is relatively low, with maximum wave heights < 2 m.

The surface water elevation measured by an instrument set up to measure waves, water levels, and currents near the Ocean Reef Marina is shown in Figure 9, alongside the timeseries of surface elevations reconstructed using a harmonic fitting of harmonic frequencies M_2, S_2, N_2, K_2, K_1, O_1, P_1, Q_1, M_4 to the measured signal, to remove the interannual variation in the water level.

[2] An assessment of the effects that nutrients (or lack-thereof) transported by this current, or the impact it may have on local water temperatures, are beyond this scope of work.

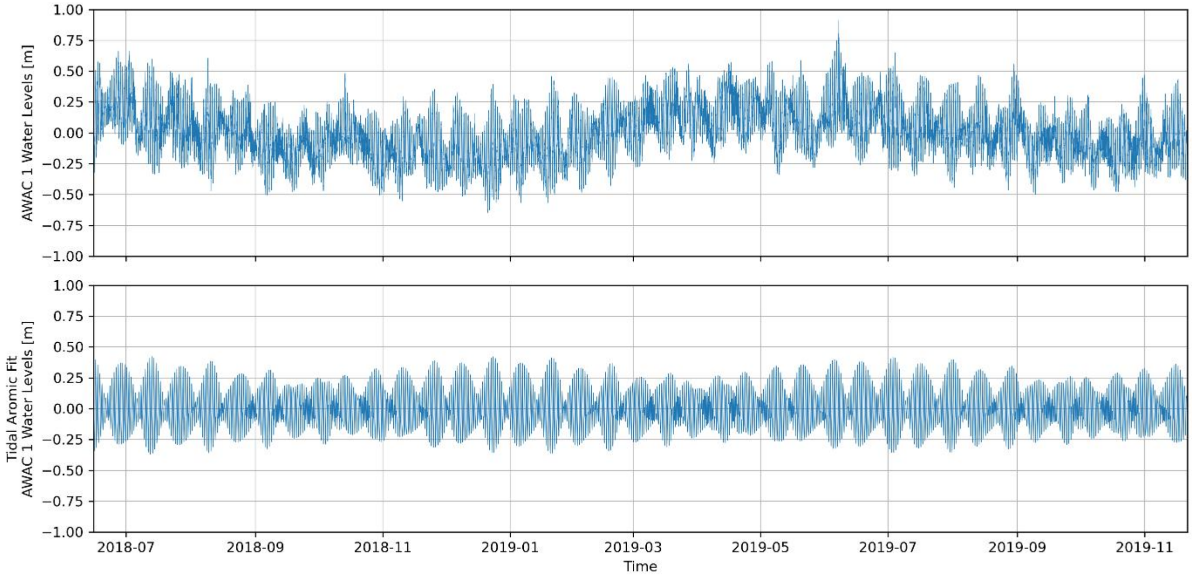


Figure 9 Measured water levels near the Ocean Reef Marina (top) and harmonic tidal fitting (bottom).

2.4. Water Circulation

At a regional scale, water circulation along the Perth offshore shelf is dominated by the LC and wind forcing. The LC is originating from the northwest and transports warm, low-salinity, nutrient-poor tropical water (Cresswell, 1991). The LC is stronger and closest to shore during the autumn and winter months, while is weaker and further offshore during the summer months. These fluctuations in the LC are mainly driven by the wind patterns changing throughout the year. Summer months are mainly dominated by south/south-westerly sea breezes which weakens and forces the LC further offshore, leaving space for seasonal wind-driven near-shoreline current known as Capes Current (Pearce and Pattiaratchi, 1999). Winter months have more consistent westerly winds pushing the LC closest to shore.

Around the Ocean Reef Marina, tides produce only weak currents of the order of 0.01 m/s, whereas the predominant driving force for currents is wind stress acting on the sea surface (Steedman and Associates, 1976; Pattiaratchi 1991), particularly when the wind intensity is >4 m/s. Pattiaratchi (1991) explained that due to the predominantly northerly wind stress, in summer, the net water movement is to the north at speeds ranging from 0.02 to 0.08 m/s, however there is no dominant direction for the currents in winter. When the Capes Current is present, nearshore flow can reach 0.3 m/s and occasionally exceed 0.5 m/s. According to Prof. Pattiaratchi, there is also a seiche motion with periods of 20 - 40 minutes that can generate cross-shore currents of up to 0.20 m/s, however these currents do not induce a net current flow and therefore are secondary to this study.

Datasets of locally measured currents were provided by the DoT for three measurement sites, as summarised in Table 5.

Table 5: DoT Current meters data period and deployment locations.

Site	Collection period	Latitude	Longitude
Ocean Reef AWAC1	June 2018 - November 2019	31.757526°S	115.721430°E
Ocean Reef AWAC2	June 2021 to July 2022	31.756520°S	115.721233°E
Hillarys Signature (SIG_HIL)	November 2021 – December 2022	31.829827°S	115.722347°E

Ocean Reef AWAC1 (Figure 10) captured pre-redevelopment, while Ocean Reef AWAC2 (Figure 11) and Hillarys Signature (Figure 12) post-redevelopment conditions, respectively. Colourmaps were used to illustrate the correlation between top and bottom layers. No information on how the layers were defined nor the number of layers was received by O2Me. Generally, higher current speeds are observed in the top layer at all sites, irrespective of the period of recording.

Monthly current roses are illustrated in Figure 13 and Figure 14 for the Ocean Reef sites. During the summer months, currents predominantly flow north and northwest, driven by the summer southerly winds. In winter, a primarily southward movement is observed; however, currents in other directions are noted, including north and west, consistent with earlier measurements by Pattiaratchi (1991).

The datasets for Ocean Reef and Hillarys currents show comparable current speeds and similar current directional patterns, despite currents at Hillarys being slightly lower. The variability in the current direction at Hillarys might be explained by sensor accuracy at low currents. Nortek was contacted for comment ([3]) and confirmed that directions associated with low current speeds of <0.05 m/s should be used with caution.

[3] O2Me meeting with Nortek, dated 9 December 2024

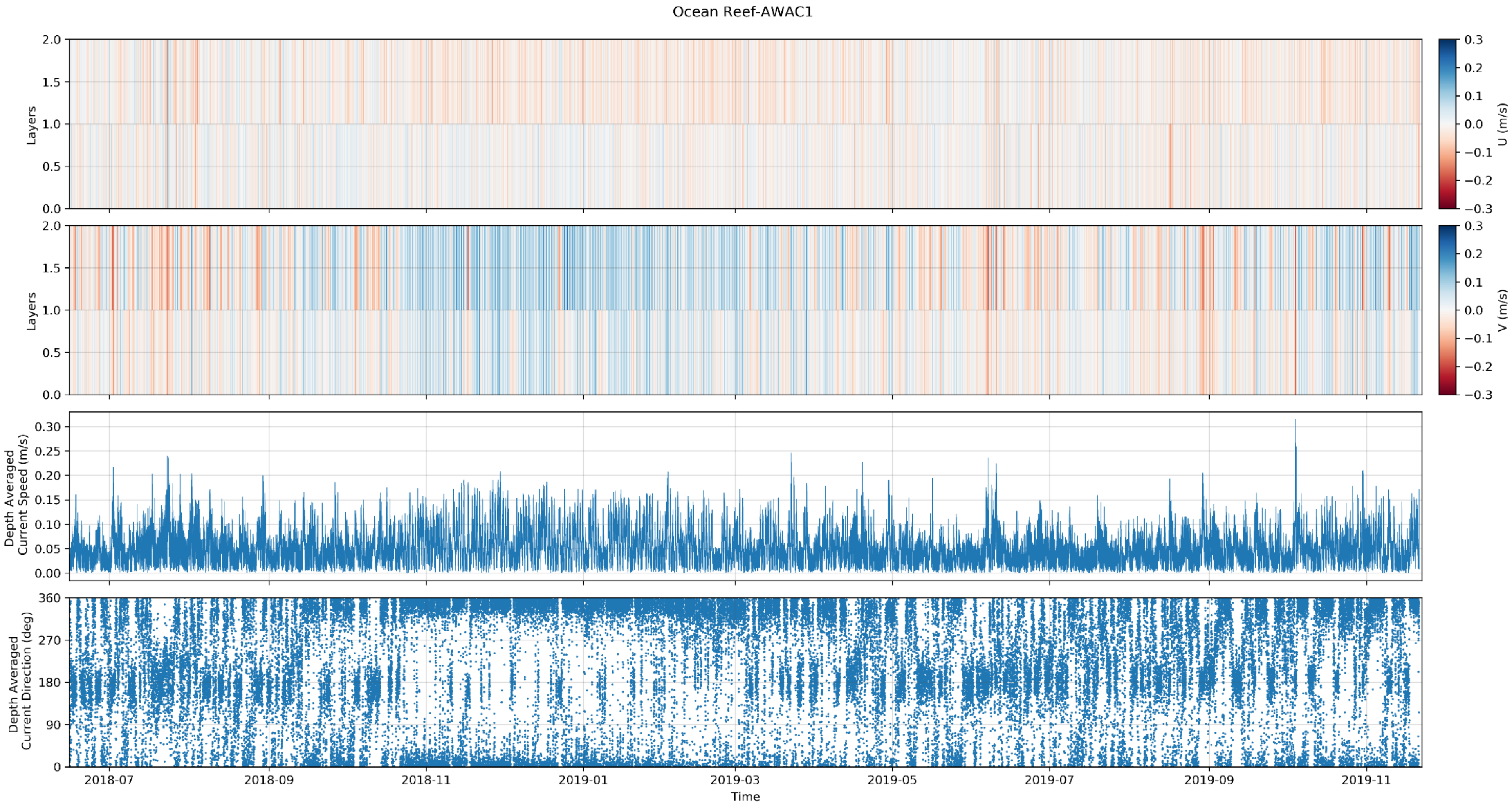


Figure 10: Ocean Reef AWAC01 Currents. Top two panels show bottom and surface along the easting and northing directions. Bottom two panels show depth averaged current speed and direction.

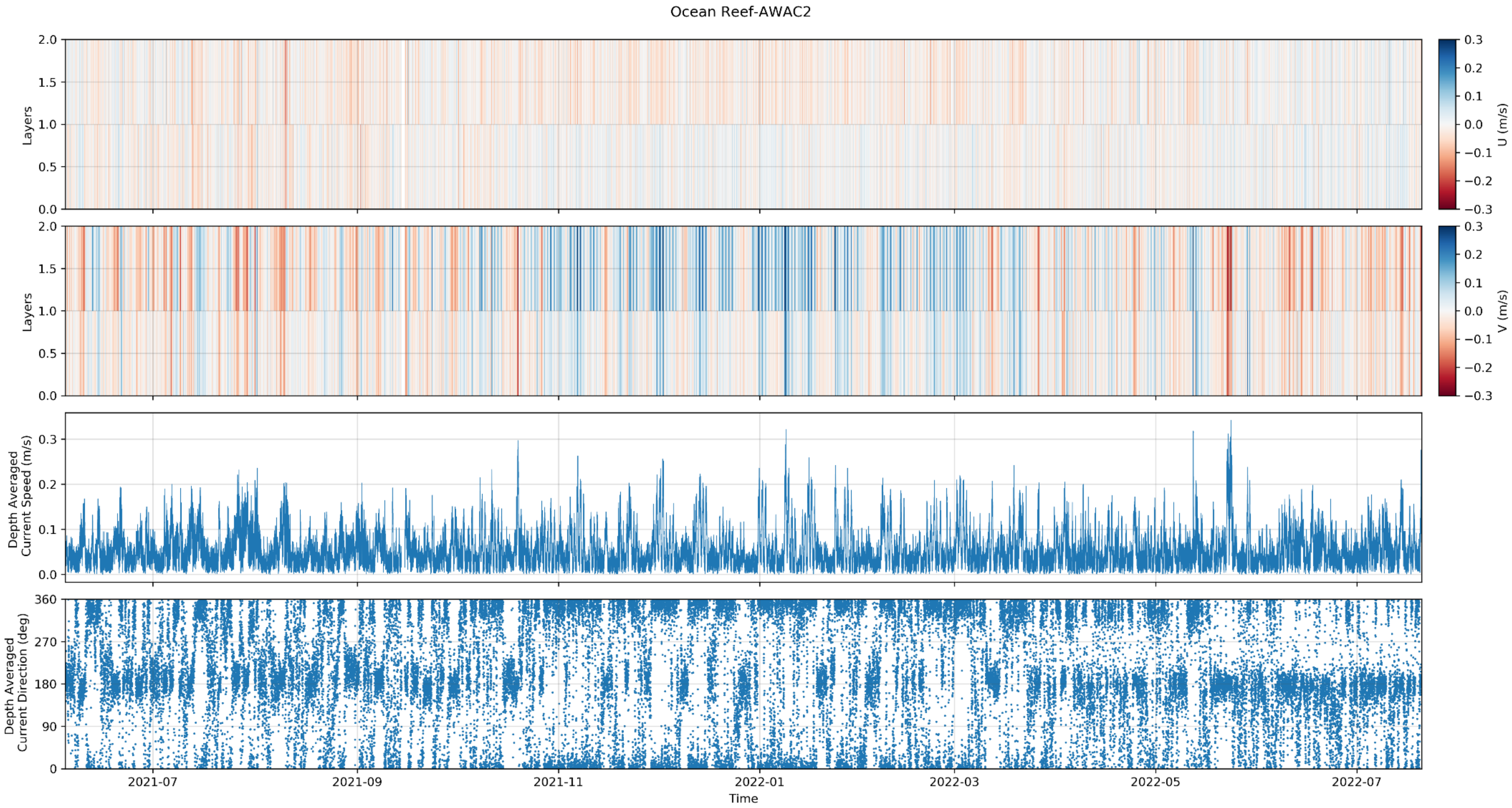


Figure 11: As per Figure 10 for Ocean Reef AWAC02.

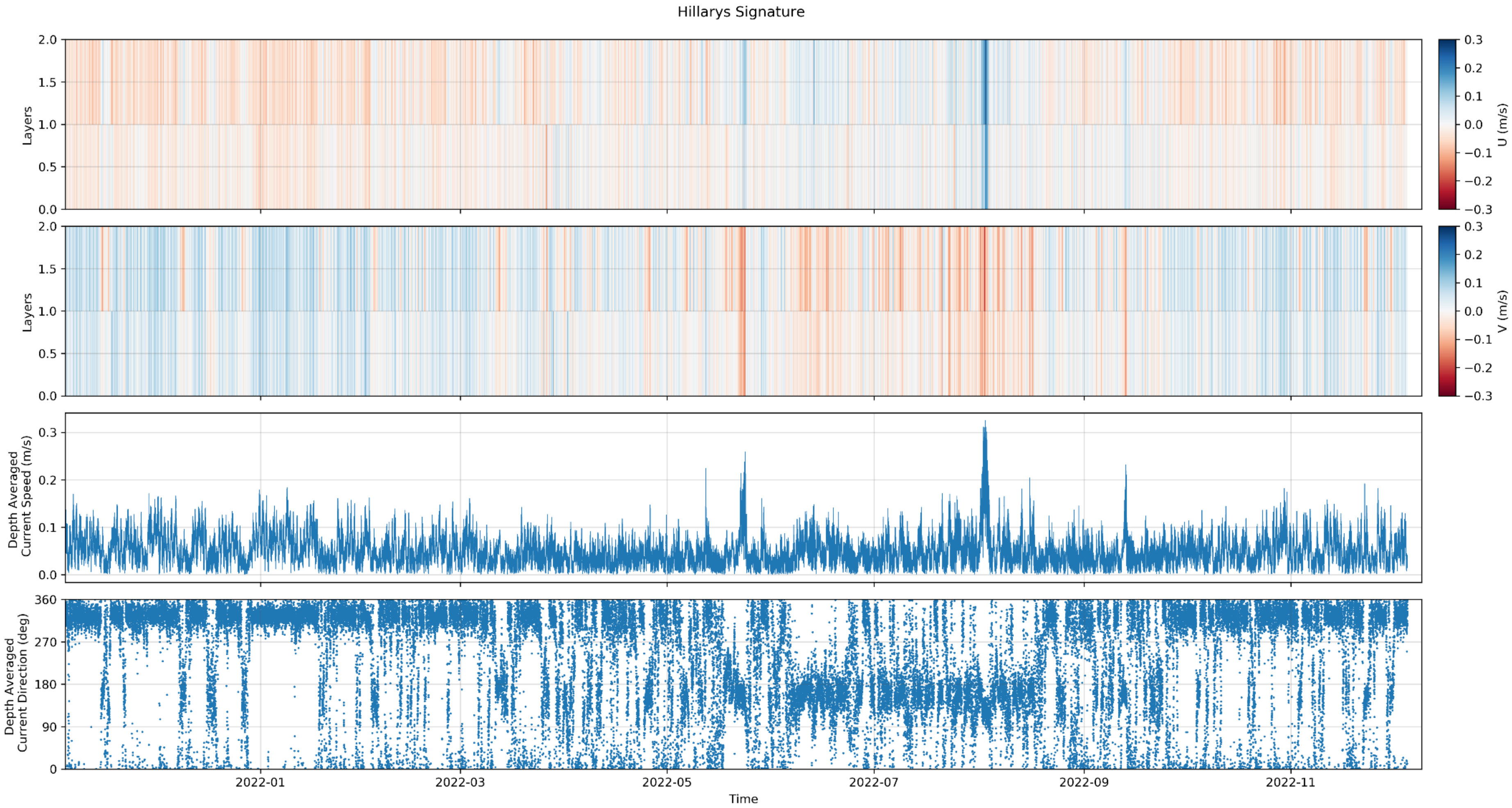


Figure 12: As per Figure 10 for Hillarys Signature.

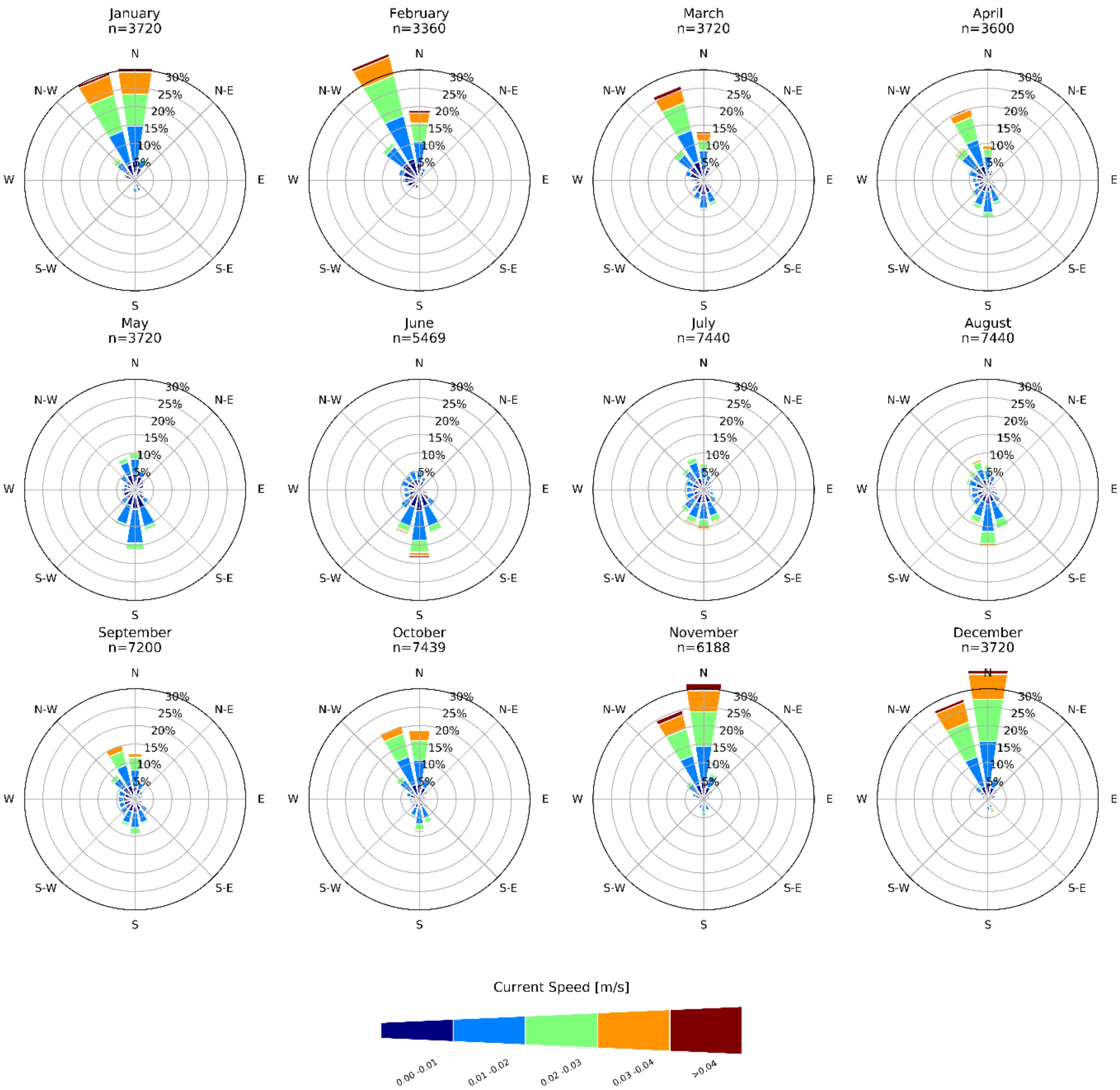


Figure 13: Monthly depth-averaged current rose at Ocean Reef AWAC 1.

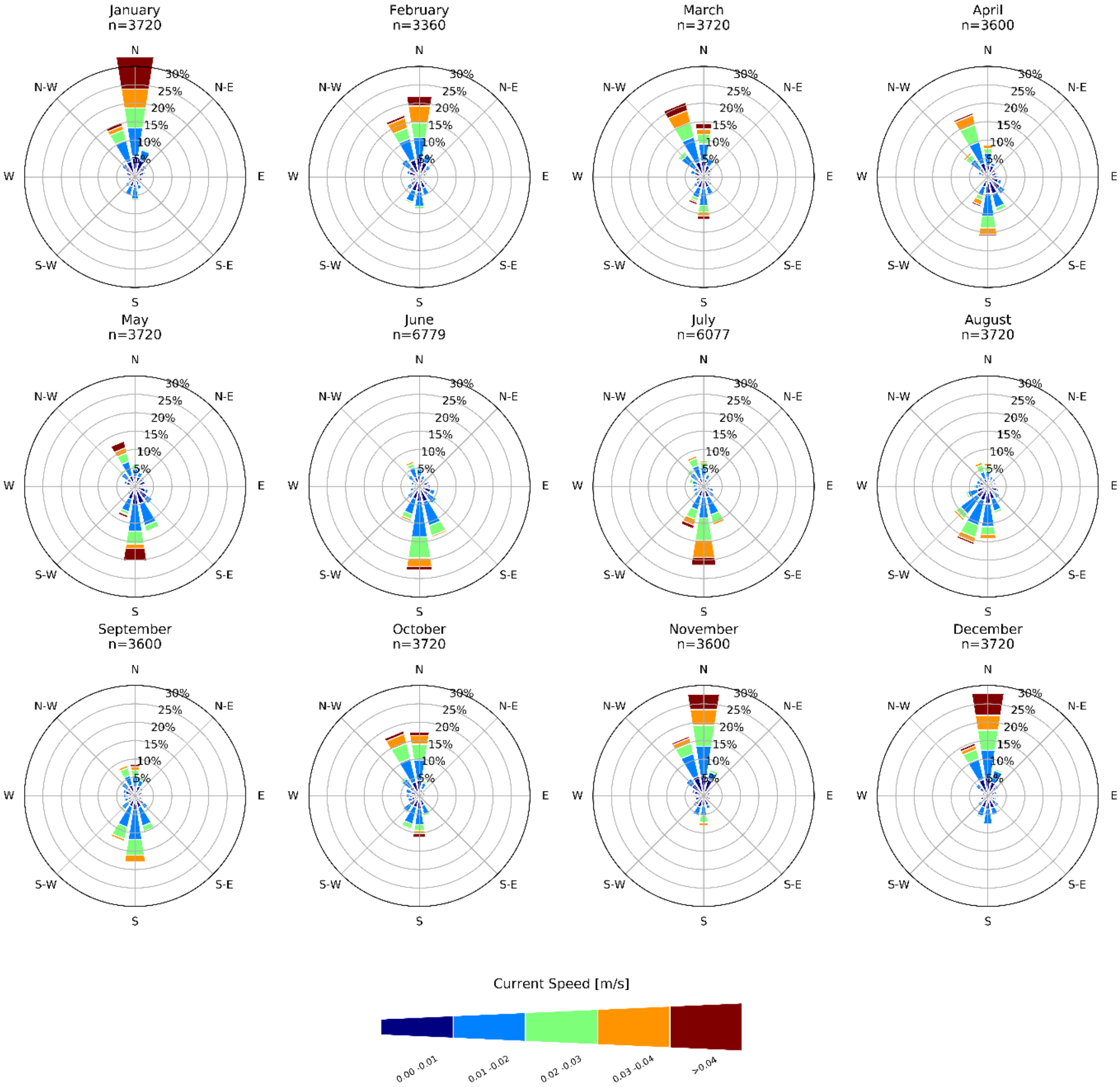


Figure 14: Monthly depth-averaged current roses at Ocean Reef AWAC 2.

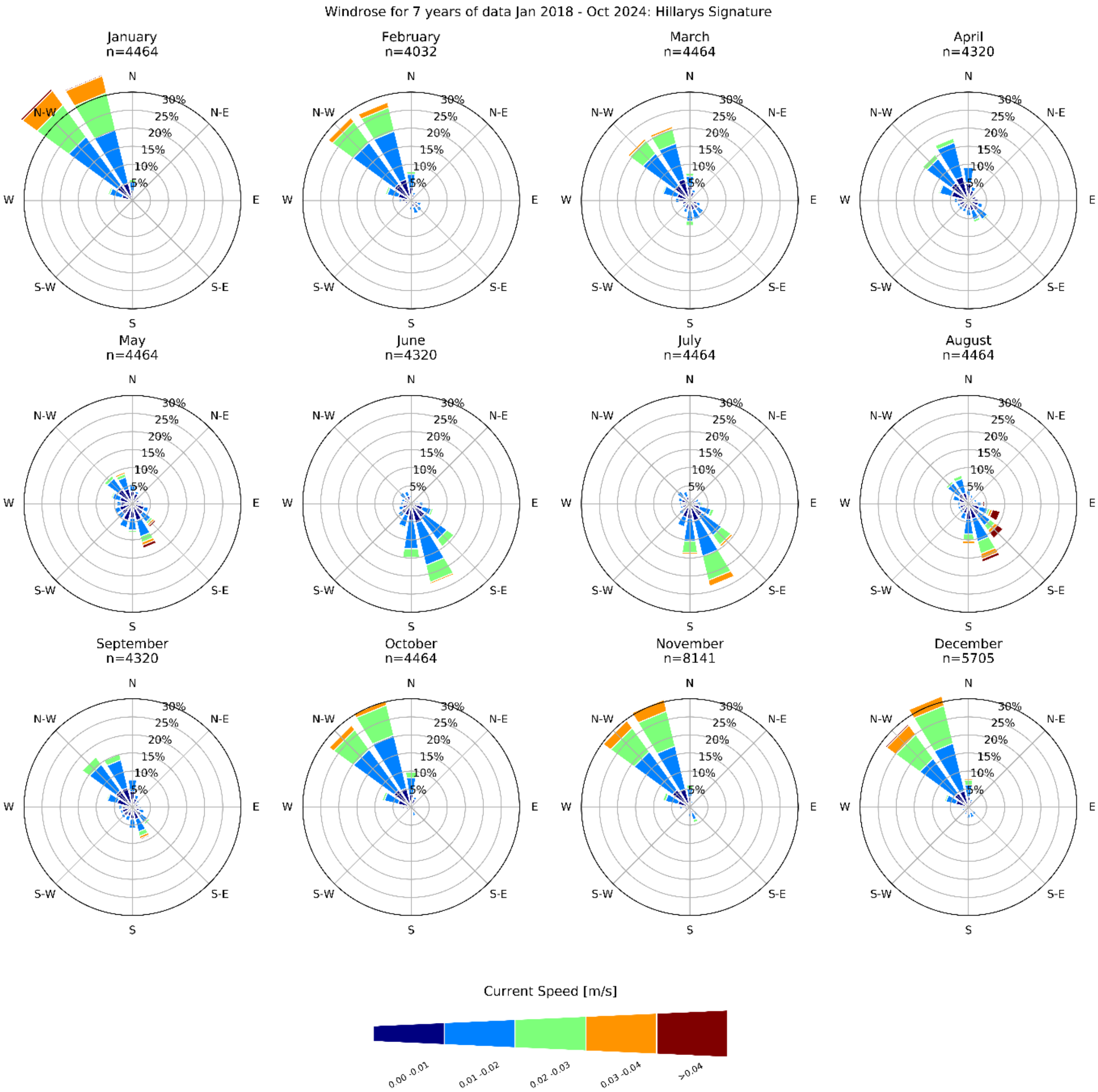


Figure 15: Monthly wind-rose plot for current field at Hillarys Signature (SIG_HIL).

2.5. Vertical Structure

Vertical profiling of water properties has been undertaken around and inside Ocean Reef Marina since the redevelopment project was first proposed (and probably earlier). Apasa (2011) reported mild-temperature stratification (<1°C between surface and bottom layers in 9 m water depth) and constant salinity ~35.1 PSU from the winter of 2011 profiles gathered between the pre-development Ocean Reef Marina and Beennyup outfall discharge site. More recently, WaterCorp collected several CTD profiles in the proximity of Ocean Reef Marina (2023-2024 summer months).

Twelve (12) CTD profiles were randomly picked from a large pool of CTD profiles collected by the WaterCorp over the summer months in 2023-2024 to characterise typical summer water temperature and salinity conditions (Figure 16) ([4]). Profiles of salinity are practically homogeneous except where the influence of the freshwater signal from the Beenyup effluent is observed. Profiles gathered in late mornings during hot summer days present mild temperature stratification of up to 1°C between surface and bottom, in agreement with earlier observations by others (e.g. Pattiaratchi 1991).

An ambient water temperature of 22.5°C and salinity of 36.2 psu may be adopted as background ambient conditions for numerical modelling.

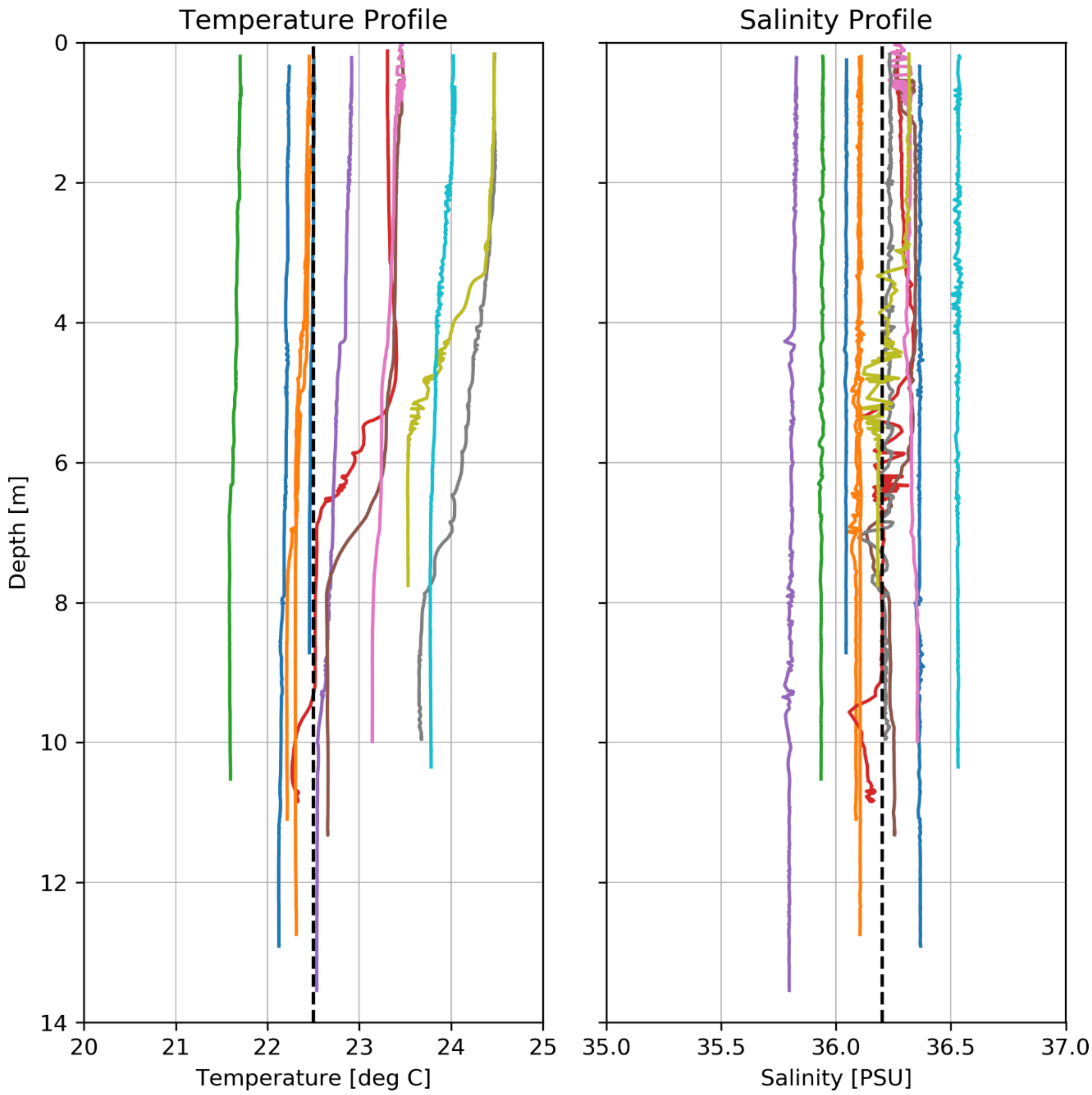


Figure 16: Sample of 12 profiles across the summer 2023-2024 profile data provided by WaterCorp.

[4] To support the discussion on water quality, depth-averaged results are provided in Section 3.3.3.1.

3. Water Quality Review

3.1. Data Sources

O2M has undertaken a review of water quality datasets with the purpose of characterising changes to the discharge stream and supporting the modelling study evaluation of whether redevelopment of the Ocean Reef Marina could have substantially altered the distribution of constituents discharged by the Beenyup effluent. The data sources provided to O2M to perform the data review included:

- WaterCorp - Perth Long Term Ocean Outlet Monitoring (PLOOM) for the Beenyup Water Resource Recovery Facility
- DWER – Recent sampling and analysis carried out in response to algal and water quality incidents.
- Department of Health – Microbial water monitoring program.

Additional data within publicly available annual reports provided on the WaterCorp website for the Beenyup Water Resource Recovery Facility was also included, where applicable (Water Corp 2024).

Proponents of developments are required to adhere to Ministerial Conditions and licence conditions applied to their project as part of the environmental approval process. These approval conditions often include the development and implementation of management and monitoring plans to ensure environmental protection is upheld. Details of the datasets from monitoring programs used in this report are provided in Table 1.

Table 1: Details of recent data sources analysed as part of investigations into water quality at Mullaloo beach

Owner	Data type	Time period
Water Corporation	PLOOM Monitoring	Jul 2022-Jan 2024
Water Corporation	PLOOM Monitoring Annual Reports	2018 - 2023
DWER	Algal and physiochemical properties	Jan 2024-Apr 2024
Department of Health	Microbial testing	Nov 2022-Mar 2024

3.2. Environmental Quality Guidelines

The guidance on marine environmental quality in Western Australia outlines an Environmental Quality Management Framework (EQMF) (EPA 2016). The framework is based on the principles and guidelines of the National Water Quality Management Strategy (NWQMS), regarding the Australian and New Zealand Guidelines for Fresh and Marine Water Quality (ANZG 2018).

The Environmental Values (EVs), Environmental Quality Objectives (EQOs) and, for the EQO 'maintenance of ecosystem integrity', Levels of Ecological Protection (LEPs) constitute the primary management objectives and represent the community's and other stakeholder's desired outcome for marine environmental quality. Environmental Quality Criteria (EQC) represent scientifically based limits of acceptable change to a measurable environmental quality indicator that is important for the protection of the associated environmental value.

The Beenyup Ocean Outfall is managed in accordance with the EQMF. The relevant EQO's which have been identified for protection of EVs around the Beenyup Ocean Outfall include:

- Maintenance of Ecosystem Integrity
- Maintenance of Seafood for Human Consumption
- Maintenance of Primary and Secondary Recreation

This review provides an overview of data collected from the Beenyup Ocean Outfall monitoring program and other data sources provided from DWER to evaluate performance relevant to the above EQOs. The context to the potential influence of the redevelopment of the Ocean Reef Marina is discussed following the data summary.

3.3. Perth Long Term Ocean Outlet Monitoring Program

WaterCorp operates the Beenyup Water Resource Recovery Facility (WRRF) in metropolitan Perth, which treats approximately 116 ML wastewater per day to produce advanced secondary TWW. The TWW is traditionally discharged to the sea through two ocean outlets, Outlet A and Outlet B, into Marmion Marine Park at Ocean Reef (Figure 1). The outlets are 1.65 km (Outlet A) and 1.85 km (Outlet B) in length and located in ~10 m of water (Figure 1). Discharge commenced from Outlet A in 1978 and Outlet B in 1992.

A monitoring program is undertaken for the facility as part of the Perth Long-term Ocean Outlet Monitoring (PLOOM) Program. Details of the monitoring program are provided in the Bennyup Ocean Outlets Monitoring and Management Plan (BMT 2023).

3.3.1. Operational Discharge Data

Flowrates and volumes discharged at the ocean outlets were provided by WaterCorp. Flowrates cover the period 1 January 2023 to 30 September 2024 while discharge volume data covers from 1 January 2021 to 30 June 2024.

Average measured flowrates are presented in Figure 17 ([5]). A spike in flow rates occurs in February 2024 which appears to be related to erroneous data recorded from flow meter #1. From the 28 January 2024 to 6 February 2024 flowmeter #1 averaged a flow rate of 2.4 L/s, compared to 496.0 L/s at flowmeter #2. Then from 6 to 26 February 2024, flowmeter #1 recorded a mean flowrate of 3,318.2 L/s compared to 502.0 L/s at flowmeter #2.

Flowrates remain relatively consistent over the longer term despite standard deviations of ~560-605 L/s. The average flowrate in 2023 was 978.9 L/s compared to 946.7 L/s for the data covered in 2024 (minus erroneous data previously mentioned).

Daily measured discharge volumes of TWW are shown in Figure 18 and summarised in Table 6. The mean discharge volume across all years was 75,168 kL, ranging from 14,998 kL on 21 October 2022 to 168,056 kL on 5 June 2023. The highest mean daily discharge volume was recorded in 2023 (84,590 kL) and lowest in 2022 (62,971 kL), while volumes from 2021 and 2024 are comparable to the overall mean.

Table 6: Summary of daily discharge volumes (kL) from 1 Jan 2021 to 30 Jun 2024

Parameter	Min	Max	Mean	Median	Mean2021	Mean2022	Mean2023	Mean2024
Volume	14,998	168,056	75,168	73,512	77,284	62,971	84,590	76,487

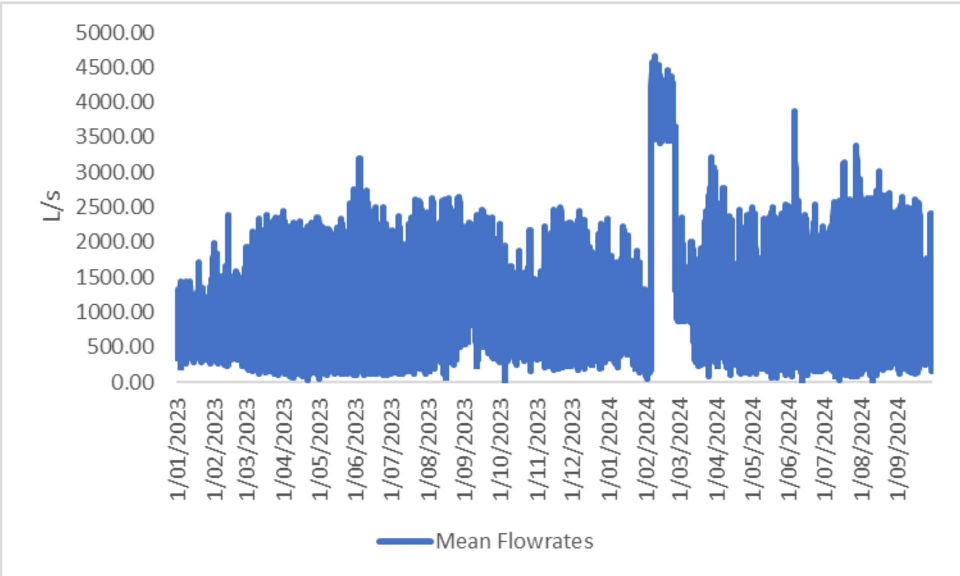


Figure 17: Average measured flowrates (L/s) from the Beenyup ocean outfall between 1 January 2023 and 30 September 2024

[5] In this section, volume rates are provided in L/s, consistent with units generally adopted in water quality studies. The Internal System of Units is used elsewhere in the report (e.g. Section 4).

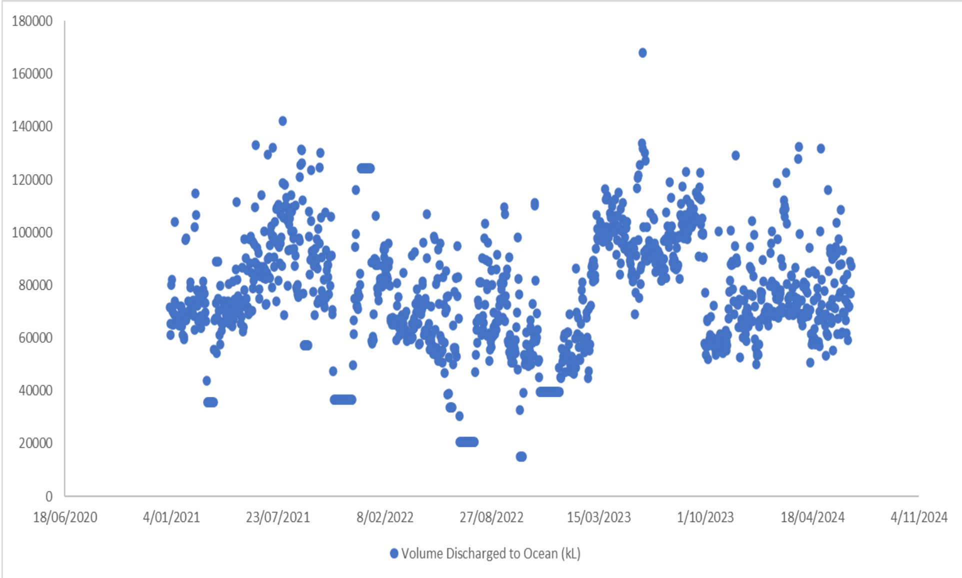


Figure 18: Daily volume (kL) of TWW discharged to ocean between 1 January 2021 and 30 June 2024

3.3.2. Treated Wastewater

This section reviews data collected from the TWW. TWW is routinely analysed for a suite of parameters:

- Nutrients (total nitrogen, ammonia, nitrate+nitrite [NOx], total phosphorus, orthophosphate)
- Microbiological contaminants (thermotolerant coliforms and Enterococci spp.)
- Bioavailable metals (arsenic, cadmium, chromium, copper, lead, mercury, nickel, selenium, silver and zinc)
- Pesticides and herbicides (organophosphate pesticides, organochlorine pesticides, triazine herbicides)
- Polyaromatic hydrocarbons
- Phthalates
- Polychlorinated biphenyls
- Benzene, toluene, ethylbenzene, and xylenes
- Petroleum hydrocarbons
- Surfactants
- Dissolved organic carbon.

Characterising changes in the wastewater stream provides context for determining the potential influence these changes have on the water quality measured in the receiving environment.

3.3.2.1. Toxicants

Samples of the TWW are collected and reported annually for the Beenyup Ocean Outfall Monitoring and Management Plan for the EQO Maintenance of Ecosystem Integrity. Samples sent to a NATA accredited laboratory for testing. The following parameters are measured as toxicants to evaluate and report on marine environmental quality:

- Dissolved metals
- Ammonia

- Pesticides
- Hydrocarbons
- Whole effluent toxicity

A summary of the toxicant in TWW data is provided in Table 7.

3.3.2.1.1. Bioaccumulating toxicants

Cadmium and mercury were identified as the potential bioaccumulating toxicants in the TWW. The EQG for these analytes require concentrations below their respective ANZG (2018) 80% species protection guidelines (Cd=36 and Mg=1.4 µg/L) prior to discharge and dilution with seawater. The concentrations of cadmium and mercury in the TWW stream have remained below the laboratory limit of reporting throughout the period of 2019-2024, and therefore have also remained well below the EQG values prior to discharge (Table 7). Therefore, there has been no detectable changes in bioaccumulating toxicants within the wastewater stream during this period.

3.3.2.1.2. Other Toxicants

Other (i.e. non-bioaccumulating) contaminants in the wastewater stream require concentrations to not exceed the ANZG (2018) 99% species protection guideline value at the LEPA boundary which forms 100 m radius from the diffuser. This is the mixing zone for initial dilution with seawater and is shown in Figure 4 and Figure 22. It is at the boundary of the LEPA where EQG must meet the requirements for the High Ecological Protection Area (i.e. the HEPA at 99% species protection) for the monitoring program.

Ammonia, copper and zinc are the only contaminants within the TWW which contain concentrations in the waste stream above the 99% species protection guideline values that require dilutions to meet the EQG at the boundary of the LEPA (Table 7). The 99% species protection guideline values are 500 µg/L, 0.3 µg/L and 7 µg/L, respectively. Changes in the concentrations of these contaminants within the effluent prior to discharge between 2018 and 2024 are shown in Figure 19. The highest concentrations for all three contaminants from this period were recorded in 2024.

Each year, modelling is undertaken to determine the average initial dilution achieved within the LEPA. Between 2018 and 2024, worst-case initial dilutions ranged from 1:134 in 2022 to 1:385 in 2024 (Table 7). The concentration at the boundary of the LEPA is calculated by dividing the TWW concentration by the worst-case initial dilutions for that year, and factoring the median background concentration of each constituent in seawater used to dilute the effluent. Despite elevated concentrations of ammonia, copper and zinc in 2024 compared to previous years, predicted concentrations were well below the EQG for all three contaminants on the boundary of the LEPA. Estimated concentrations at the LEPA boundary between 2018 and 2024 ranged from 3.6-8.7 µg/L, 0.1-0.2 µg/L and 0.4-0.6 µg/L for ammonia, copper and zinc, respectively (Table 7).

Results

3.3.2.1.3. Total Toxicity of the Mixture

While the contaminants described above meet their ANZG (2018) default guideline values (DGVs) on the boundary of the LEPA, these toxicants do not occur in isolation and their combined presence is assessed by calculating the total toxicity of the mixture (TTM). This has been performed in accordance with ANZG (2018) assuming that the synergistic effect of the toxicity is additive (i.e. the sum of the toxicity of the individual components). This calculation can be summarised as the sum of the contaminants (i.e. ammonia, copper and zinc) divided by their respective EQG value. If the TTM exceeds 1 (i.e. is greater than the sum of the EQG values), the mixture has exceeded the collective water quality guideline value. The TTM results based on ammonia, copper and zinc concentrations on the boundary of the LEPA calculated between 2018 and 2024 are shown in Table 7 and range between 0.42 and 0.76. Highest TTM results were recorded in 2022 although there is no evidence of increasing or decreasing trends and data is variable in nature. No results have been recorded to exceed 1 in any year.

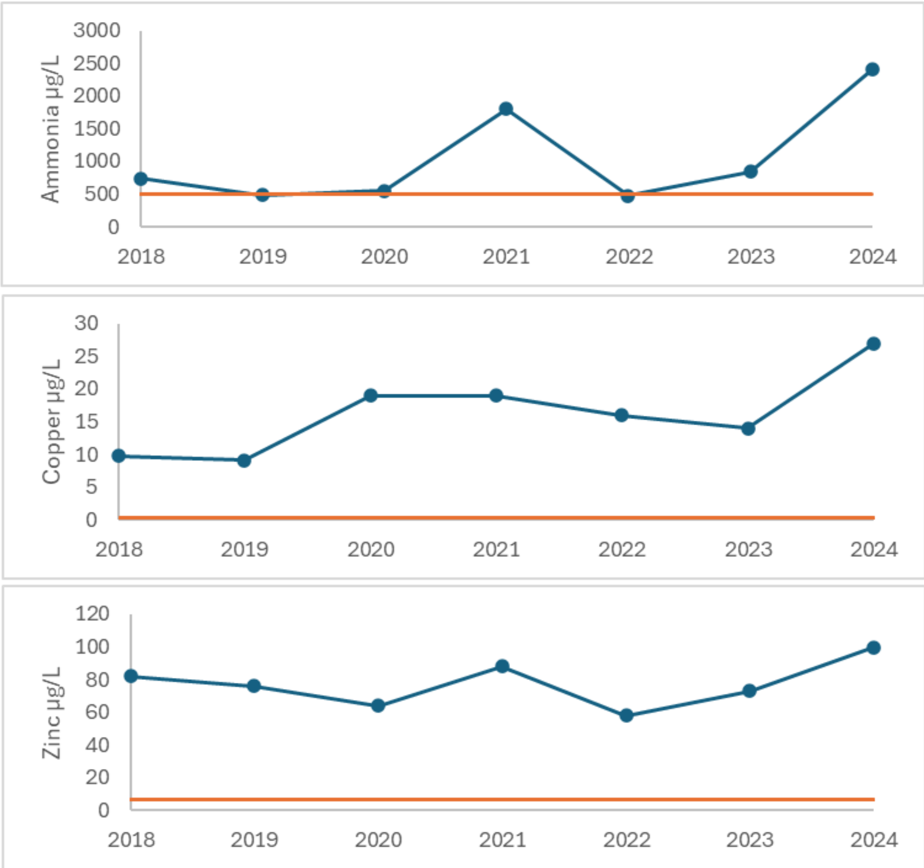


Figure 19: Changes in measured ammonia, copper and zinc concentrations in the effluent prior to discharge between 2018 and 2024. The EQG to be met at the LEPA boundary is shown as the orange line.

Table 7: Ocean Reef treated Wastewater Stream. Contaminants that exceed the EQC are highlighted and estimated concentration on the LEPA boundary shown in brackets

Group	Toxicant (µg/L)	EQC	2018	2019	2020	2021	2022	2023	2024
Nutrients	Ammonia	500	**740 (3.6)**	490 (4.4)	**550 (3.7)**	**1800 (8.7)**	480 (3.6)	**840 (2.8)**	**2400 (7.7)**
Dissolved metals	Cadmium	36		<0.1	<0.1	<0.1	<0.1	<0.1	<0.1
	Chromium (Cr VI/III)	0.14/7.7	1.4	<1	2.1	1.2	<1	<1	1.4
	Copper	0.3	**9.8 (0.2)**	**9.1 (0.1)**	**19 (0.2)**	**19 (0.2)**	**16 (0.1)**	**14 (0.1)**	**27 (0.2)**
	Lead	2.2	<1	<1	<1	<1	<1	<1	<1
	Mercury	1.4		<0.1	<0.1	<0.3	<0.05	<0.05	<0.05
	Nickel	7	5.9	2.3	3.6	2.6	2.3	3.2	4.8
	Silver	0.8	<1	<0.8	<0.8	<0.8	<0.8	<0.8	<0.8
	Zinc	7	**82 (0.8)**	**76 (0.6)**	**64 (0.4)**	**88 (0.5)**	**58 (0.4)**	**73 (0.2)**	**100 (0.4)**
Organophosphate pesticides	Chlorpyrifos	0.0005	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1
Organochlorine pesticides	Endrin	0.004	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001
	Endosulfan Sulfate	0.005	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001
BTEX	Benzene	500	<1.0	<1.0	<1.0	<1.0	<1.0	<1.0	<1.0
PAHs	Naphthalene	50	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
	Benzoperylene	50	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
Worst-case initial dilutions		Na	1:351	1:168	1:246	1:249	1:134	1:303	1:385
Total toxicity of the mixture		1	0.42	0.54	0.59	0.61	0.76	0.49	0.58

3.3.2.1.4. Whole effluent toxicity

Whole effluent toxicity (WET) testing refers to the aggregate toxic effect to aquatic organisms from all pollutants contained in TWW. WET test methods consist of exposing living aquatic organisms (plants, vertebrates and invertebrates) to various concentrations of a sample of the TWW under laboratory conditions. The Beenyup Ocean Outfall uses the sperm and egg fertilisation success in sea urchins (*Heliocidaris tuberculata*) exposed to salt adjusted dilutions (1.0, 1.6, 3.1, 6.3, 12.5, 25, 50 and 100%) of TWW diluted with filtered seawater. The test results calculate a No Observed Effect Concentration (NOEC), which describes the highest wastewater concentration where no significant effects are observed. A significant effect is determined by comparing the sea urchin fertilisation success rate in effluent diluted samples against that recorded using an artificial sea salt (brine) control sample.

The NOEC results for WET testing are shown in Figure 20. The WET testing is typically undertaken in March, July and November each year. The NOEC results were typically higher at between 50-100% in 2017-2019 compared to between 25-50% between 2019-2024, indicating toxicity of the discharge effluent increased from 2019, but this would not correlate with recent observed events.

The minimum NOEC was recorded in July 2022 at 12.5% of effluent. The results indicate a minimum of up to eight dilutions will be required for the effluent to no longer present a toxic risk. Given the worst-case estimated dilutions within the LEPA ranged between 1:134 and 1:385, typical dilutions achieved at the LEPA boundary will be far greater than the minimum number of dilutions required to achieve the NOEC. This may be interpreted as any potential risk of toxic effects from the discharge would likely occur near the outfall and is very unlikely to disperse as far as the LEPA.

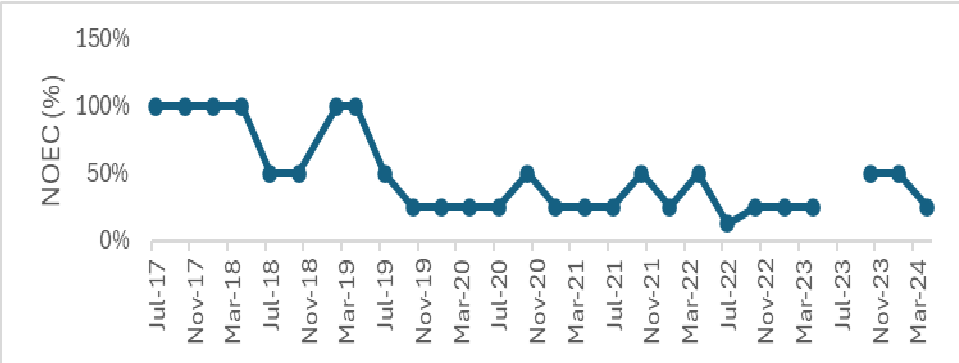


Figure 20: No observed effects concentrations from Sea Urchin fertilisation success WET testing between 2017 and 2024

3.3.2.2. Physico-chemical

Physico-chemical data measured on the TWW stream prior to discharge on a periodic basis as part of the monitoring program includes parameters shown in Table 8. These parameters in the waste stream are not reported online and data was provided from WaterCorp for the period 5 July 2022 – 4 June 2024, although dates can vary depending on the parameter.

Daily pH values remained relatively constant from 1 January 2023 to 21 October 2024, averaging 7.2 and ranging between 6.37 and 7.83. There was minimal difference in the mean pH values between 2023 and 2024. Alkalinity, which is a measure of the acid-neutralizing capacity of water, recorded a mean of

158.2 mg/L and results were variable ranging from 75 mg/L to 290 mg/L. The buffering capacity of the effluent was slightly higher in 2024 (166.5 mg/L) compared to 2023 (152.5 mg/L).

Weekly readings for conductivity and triannual samples for TDS indicate mesohaline to polyhaline effluent. Conductivity averaged ~20,000 µS/cm and ranging from 11,460 to 43,000 µS/cm, while TDS averaged 1340 mg/L, ranging from 670 to 1830 mg/L. Results indicate the TWW was slightly more brackish in 2024 compared to 2023.

Turbidity readings are collected tri-annually and suspended sediment is sampled daily. The effluent discharged was relatively turbid averaging 8.4 NTU and ranging from 4.6 to 14 NTU while suspended solids averaged 30.2 mg/L, ranging from 5 to 90 mg/L. Suspended sediment results indicated discharges in 2024 comprised higher concentrations (mean = 39.2 mg/L) than compared to 2023 (mean = 23.5 mg/L).

Effluent BOD and COD concentrations are sampled weekly which generally represent treated polluted water sources, with BOD averaging 10.3 mg/L and ranging between 1 and 32 mg/L, while COD values averaged 77.1 mg/L, ranging from 28.8 to 136.0 mg/L. The results for BOD/COD were slightly higher in 2024 than 2023, averaging 11.2 mg/L versus 9.5 mg/L and 91.3 mg/L versus 64.6 mg/L.

Table 8: Summary of physico-chemical measurements of the TWW between Jan 2023 and Oct 2024

Parameter	Units	Min	Max	Mean	Median	Mean2023	Mean2024
pH		6.37	7.83	7.23	7.22	7.23	7.22
Alkalinity	mg/L	75	290	158.2	157	152.5	166.5
Conductivity	µS/cm	11,460	43,000	20,170	19,520	18,480	22,250
Total dissolved solids (TDS)	mg/L	670	1830	1340	1500	1403.3	1292.5
Turbidity	NTU	4.6	14	8.4	6.6	8.0	8.8
Suspended solids	mg/L	5	90	30.2	25	23.5	39.2
Biochemical oxygen demand (BOD)	mg/L	1	32	10.3	9	9.5	11.1
Chemical oxygen demand (COD)	mg/L	28.8	136	77.1	73.7	64.6	91.3

3.3.2.3. Nutrient enrichment

Nutrients recorded from the wastewater stream is shown in Table 9. These nutrients are expectedly higher than naturally occurs in ocean waters. Data for nutrients were provided between the period July 2022 - October 2024.

Total nitrogen (TN) and total phosphorus (TP) were collected weekly. Mean TN was 30.1 mg/L and ranged from 15 mg/L to 58.7 mg/L, while TP recorded a mean of 12.6 mg/L, ranging from 5.8 mg/L to 26.5 mg/L. Mean concentrations in 2024 were higher than in 2023, although less than 2022. The TN and TP results are presented graphically in Figure 21 to investigate trends in nutrient concentrations in relation to the *Trichodesmium* algal bloom events at Mullaloo Beach. Firstly, the figure shows a close relationship between trends in TN and TP concentrations (i.e. elevated concentrations of TN typically correlate with elevated TP). Nutrient data indicates a couple of spikes in TN and TP occurred between January and March 2024 reaching 58.7 mg/L and 18 mg/L, respectively.

Nitrogen recorded was primarily comprised of nitrate (NO_3), averaging 22.2 mg/L. Comparably, nitrite (NO_2) and ammonium (NH_4) recorded means of 0.3 mg/L and 2.0 mg/L, respectively. Total kjeldahl nitrogen (TKN) recorded comparably higher concentrations, representative of organic nitrogenous organic compounds in the effluent, with a mean of 7.1 mg/L. Nitrate and nitrite average concentrations were slightly higher in 2023, similar to TN, although TKN and ammonium concentrations were slightly elevated in 2024 above concentrations recorded in both 2022 and 2023. The results for filterable reactive phosphorus indicate most of the TP recorded in the effluent is comprised of dissolved bioavailable orthophosphate available to be utilised by organisms, although concentrations in 2024 were lower than 2023.

In reviewing Department of Environment and Conservation's preliminary review findings of the impact of the Ocean Reef Outfall on the Marmion Marine Park, reported values for average daily nutrient load are currently approximately half of the levels observed between 1995-2010 (DEC 2011). This is attributable to technological and infrastructure related improvements in treating and testing of wastewater for ocean disposal and reduce the volume of nitrogen reaching the marine environment.

Table 9: Summary of nutrient measurements (mg/L) of the TWW between Jul 2022 and Oct 2024

Parameter	Minimum	Maximum	Mean	Median	Mean2022	Mean2023	Mean2024
TN	15	58.7	30.1	29	36	27.3	32.7
NO_3	9.1	51.6	22.2	21.2	-	21.4	23.2
NO_2	0	0.9	0.3	0.3	-	0.3	0.3
NOx	10.75	43.8	24.3	23.8	28.4	21.8	24.1
TKN	2.6	29	7.1	6.4	7.4	5.7	8.7
NH_4	0	11.7	2.0	1.7	2.1	1.7	2.4
TP	5.8	26.5	12.6	12.0	15.3	11.3	13.8
FRP	8.1	16.4	12.3	12.6	-	13.8	12.6

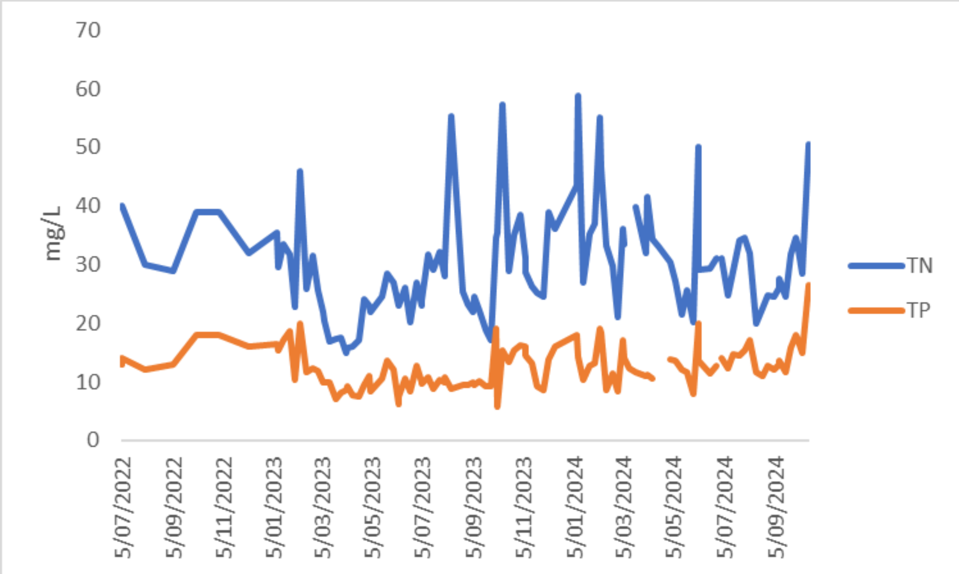


Figure 21: Measurements of total nitrogen (TN) and total phosphorus (TP) in TWW prior to discharge from the outfall between July 2022 and October 2024

3.3.2.4. Cations and Anions

Samples of the TWW for select anions, cations and alkalinity are typically undertaken triennially and results are presented in Table 10.

Anions measured include chloride, bicarbonate and cyanide. Mean concentrations recorded 388.3 mg/L, 203.3 mg/L and 0.04 mg/L, respectively. Slightly higher chloride and bicarbonate levels were recorded in 2023, while cyanide concentrations were slightly higher in 2024. The cation sodium averaged 310 mg/L and was slightly higher in 2023. There are many anions already listed, such as the nitrate and nitrite, although data is not available for some common anions used in wastewater treatment such as phosphate and sulfate. Mean concentrations for the cations calcium and sodium were 60.4 mg/L and 310 mg/L, respectively. Results indicate the concentration of major cations reduced slightly in 2024, similar to findings of anions. Cations such as heavy metals have been listed already, although data is not provided for some cations commonly used in wastewater treatment such as magnesium, iron and manganese. However, it is assumed the intent of monitoring is to measure process chemicals and not determine ionic balance using major ionic substances measured in the sample.

Table 10: Summary of ionic compound and alkalinity measurements (mg/L) of the TWW between Jan 2023 and Oct 2024

Parameter	Minimum	Maximum	Mean	Median	Mean2023	Mean2024
Chloride	220	580	388.3	392.5	426.7	407.5
Bicarbonate	110	260	203.3	215	233.3	200
Cyanide	0.006	0.11	0.04	0.02	0.02	0.06
Calcium	34	85	60.4	57	66.3	64.5
Sodium	160	440	310	325	315	305

3.3.2.5. Pathogenic Bacteria

Escherichia coli in TWW was sampled on four occasions between January 2023 and July 2024. A summary of results from samples tested are provided in Table 11. Mean *E. coli* concentrations recorded was 11,370 mg/L, ranging from 110 mg/L to 20,000 mg/L. The mean in 2023 of 10,130 mg/L was higher than recorded in 2024 at 7,055 mg/L. The maximum concentration was recorded in July 2023, the lowest concentration was recorded on 9 January 2024.

Table 11: Summary of *E. coli* concentrations (mg/L) in the TWW between Jan 2023 and Jul 2024

Parameter	Minimum	Maximum	Mean	Median	Mean2023	Mean2024
Escherichia coli	110	20,000	11,370	14,000	10,130	7,055

3.3.3. Water Quality Monitoring

Monitoring of receiving waters has been undertaken as part of the Beenyup Ocean Outfall Monitoring and Management Plan. Fortnightly sampling has been conducted between December and March each year (i.e. eight occasions per year), which involves collection of the following physico-chemical parameters:

- Salinity
- Temperature
- Dissolved oxygen
- Irradiance
- Nutrients
- Chlorophyll-a

Sampling is conducted during the summer to avoid potential confounding effects of riverine and groundwater discharge. During each survey, the gradient monitoring design involves collecting samples at increasing intervals along a transect from the outfall (0 m, 100 m, 350 m, 1000 m, 1500 m) in the direction of the current (ORT sites) and at four reference sites (ORR sites). The sites where sampling has been undertaken in summer 2023/24 are presented in Figure 22 to help visualise the monitoring program.

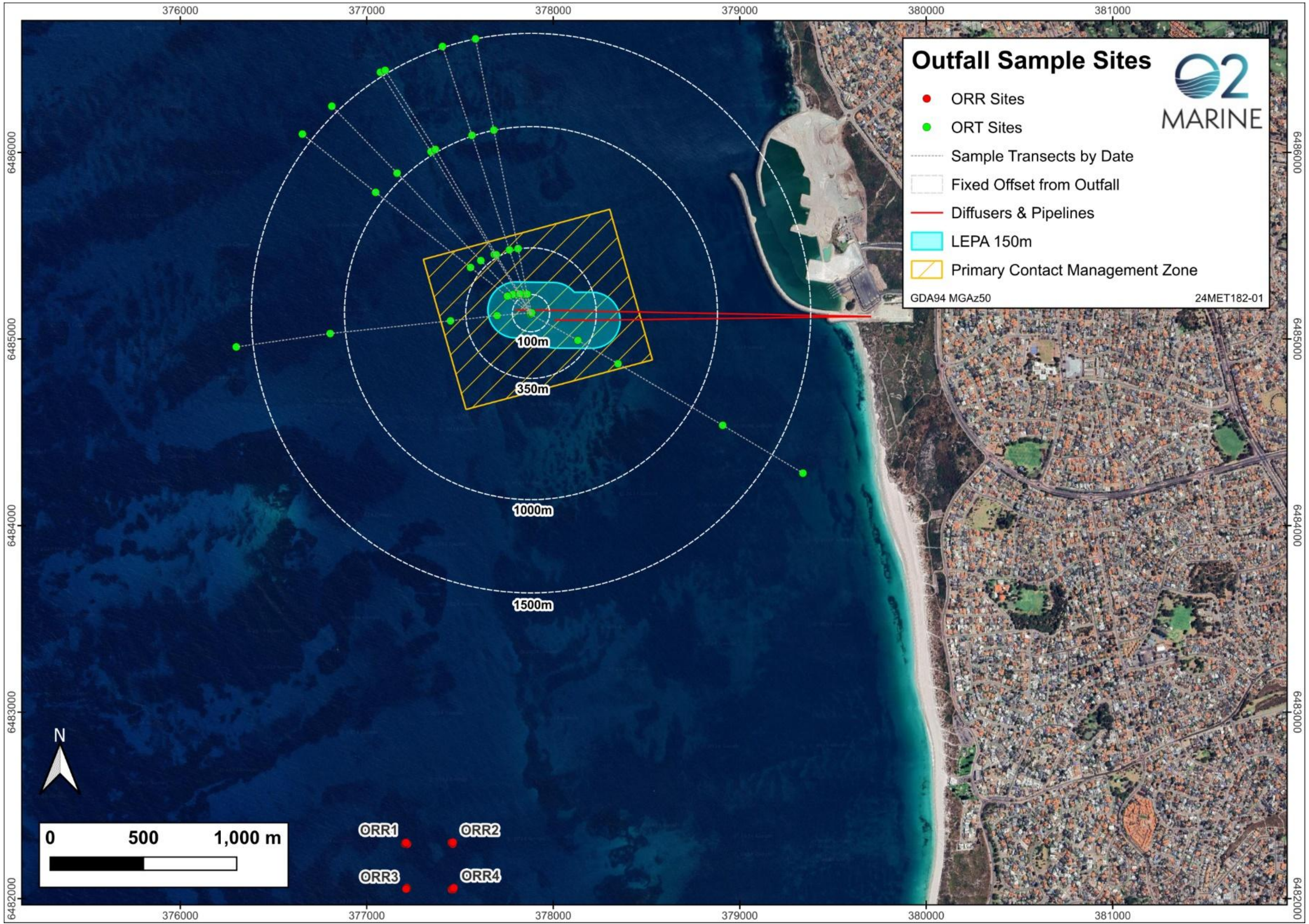


Figure 22: Water quality monitoring locations

3.3.3.1. Physico-chemical

Salinity, temperature and dissolved oxygen profile measurements are collected within the physico-chemical parameters monitored as part of the Beenyup Ocean Outfall Monitoring and Management Plan. Depth-averaged results across all sites for these parameters between 6 December 2023 and 19 March 2024 are shown in Figure 23. The results generally present relatively consistent values across all sites, including impact monitoring locations at increasing distance from the outfall and four reference locations.

Salinity across all sites and surveys averaged 36.2 psu, ranging from 34.2 psu to 36.6 psu. The data indicates salinity values are typically homogenous across sites and surveys. The most apparent exception was the minimum salinity concentration of 34.2 psu recorded from the outfall (ORT_0m) on 19 December 2023, which is compared to minimum concentrations from other sites ranging from 35.9 psu to 36.1 psu. The minimum difference to the next lowest value from another site on this survey was ~1.7 psu, which is evidence of freshwater TWW discharge which was no longer detectable 100 m from the outfall. Comparably, the difference between minimum values recorded from the outfall site and other sites on 23 January was 0.4 psu and <0.1 psu in remaining surveys.

The EQG for salinity compares the summer median concentration at 0.5 m below the water surface from an individual HEPA site down current of the LEPA boundary to remain within the 20th and 80th percentile of reference site salinity over the same period. The results from the 2023/24 summer indicate that the median concentrations from all sites, including within the LEPA at the outfall (ORT_0m) were within the reference 20th and 80th percentiles (BMT 2024). From reports available online, the EQG was only exceeded in 2019, for which the calculated median from the outfall (ORT_0m) and 100 m (ORT_100m) sites were below the EQG.

Temperature across all sites and surveys averaged 22.8°C, ranging from 20.4°C to 24.7°C. Temperatures reached maximum concentrations >24°C at the end of January 2024, then dropped to minimum concentrations at the start of February 2024 (Figure 23).

The mean range in temperature between sites within surveys was 1.2°C. Homogenous results (<1°C) were recorded on 19 December 2023, 12 January 2024 and 19 March 2024, while the temperature recorded from 16 February 2024 was particularly variable (2.1°C) between sites. The transect results for salinity, temperature and dissolved oxygen for this survey are shown in Figure 24. Notably, this transect was oriented towards the south-east towards Mullaloo Beach due to light winds (0-7 kn) from the north-east/north-west, which is an atypical direction for winds and currents over the summer months. The survey results indicate water temperature increased towards the shoreline at Mullaloo Beach in shallower water, although temperatures were comparably lower at reference sites further offshore. Similar trends are observed in salinity which is influenced by temperature, while dissolved oxygen concentrations were slightly higher at reference sites. The remaining four surveys recorded a difference of 1.4°C between sites which may indicate natural variability, although surveys in late January/early February suggest discharge TWW was slightly elevated in temperature which decreased with distance from the outfall.

Dissolved oxygen across all sites and surveys averaged 94.7% saturation, ranging from 76.4% to 109.8%. Dissolved oxygen concentrations were typically higher among impact monitoring sites compared to reference locations. For example, the mean of depth-averaged impact sites across surveys was 98.5% compared to 93.1% across reference locations.

The difference in dissolved oxygen concentrations recorded during surveys averaged 16.7%. The smallest difference of 12% occurred on 16 February 2024 where concentrations at the reference sites were slightly higher than impact sites, for which all depth-averaged results were supersaturated from 103.9% to 106.5%. The largest discrepancy in dissolved oxygen concentrations recorded was 28.7% during the next survey on 8 March 2024. Depth-averaged dissolved oxygen concentrations from this survey ranged from 91.7% to 102.8%, with the highest values at the outfall and lowest at reference sites.

The EQG for dissolved oxygen is not met when the summer median concentration in bottom waters (0–0.5 m above the sediment surface) at HEPA sites is <90% saturation for a duration of six weeks. While this was not exceeded during the recent 2023/24 summer, results recorded at reference sites were <90% for a duration of six weeks which has consistently been recorded during historical sampling (BMT 2024). The EQG has not been met for HEPA sites historically on two occasions, in the summers of 2010/11 and 2022/23. The exceedance in 2022/23 fell below 90% saturation at the ORT-1500 m site during the period between 17/01/2023 and 20/02/2023. On both these occasions, dissolved oxygen remained above the environmental quality standard (EQS) which requires dissolved oxygen to drop to <60% for a duration of six weeks (BMT 2024). Results indicate dissolved oxygen concentrations surrounding the outfall may be elevated from TWW discharge, although may be attributable to natural variation in dissolved oxygen between the two locations due to differences in physical influences.

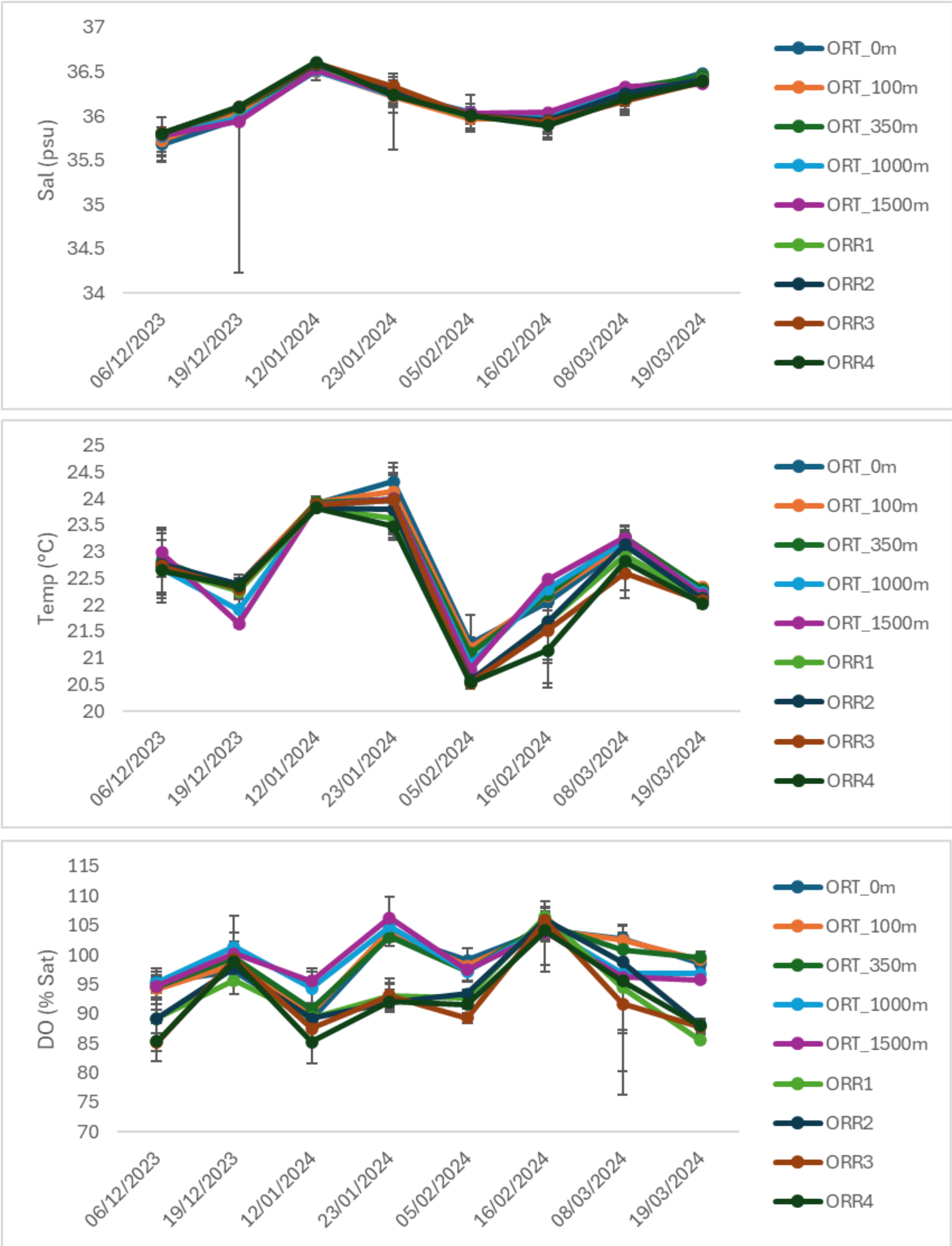


Figure 23: Depth-averaged salinity, temperature and dissolved oxygen (±maximum and minimum) results recorded during monitoring across all sites undertaken between 6 December 2023 and 19 March 2024.

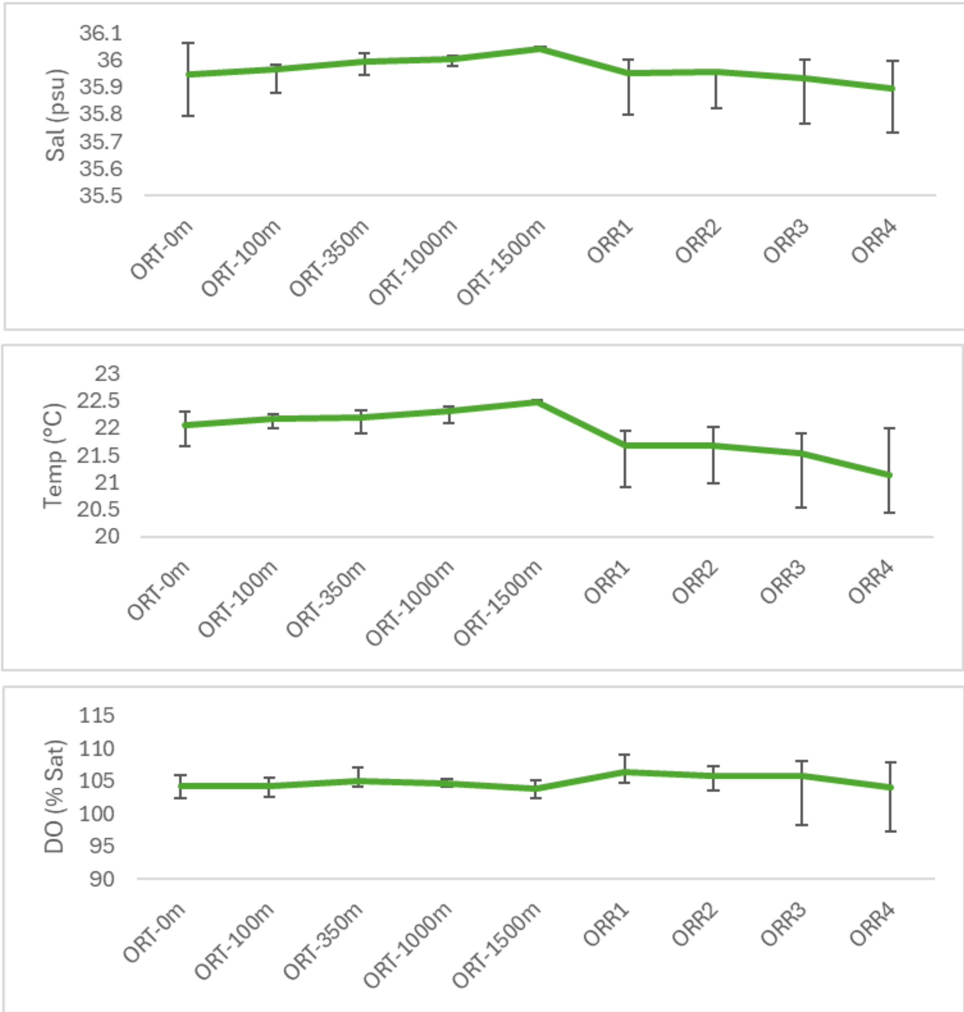


Figure 24: Depth-averaged salinity, temperature and dissolved oxygen (±maximum and minimum) recorded from each site sampled on 16 February 2024

Irradiance measurements are undertaken for the Beenyup Ocean Outfall Monitoring and Management Plan using one sensor positioned 1 m below the surface and a second sensor 7 m below the surface and the light attenuation coefficient (LAC) is calculated. Results are presented in Figure 25.

The LAC across all sites and surveys averaged 0.10 Log_{10}/m, ranging from 0.07 Log_{10}/m to 0.13 Log_{10}/m. The results exhibit typically higher LAC values for the outfall site (ORT_0m) and at 100 m distance along the transect, compared to all other sites. For example, the mean of these sites across surveys were 0.12 Log_{10}/m and 0.11 Log_{10}/m, respectively, compared to between 0.09 Log_{10}/m to 0.10 Log_{10}/m across remaining sites. The results generally indicate the discharge has some effect on light attenuation within the water column in the vicinity of the outfall, presumably associated with organic particulates released from the outfall. Highest LAC measured at the outfall site was recorded on 8 March 2024, while lowest values were recorded on 16 February 2024. These lowest LAC values represent the only transect where concentrations measured at control sites were higher than that measured from the outfall.

The EQG monitored as part of the Beenyup Ocean Outfall was not met in 2023/24 summer where the median LAC in the HEPA (0.098 Log_{10}/m) was above the 80th percentile of historical reference site data

(0.094 Log10/m, BMT 2024). However, the EQS of two consecutive years has not been triggered. BMT (2024) describe the exceedance of the EQG was due to an increase in background light attenuation in 2023/24, as the 80th percentile of reference site data for the corresponding period (0.11 Log10/m) was above the historical background and the median of HEPA sites in 2023/24. The LAC EQG has exceeded across five additional years historically, recorded between 2007/08 and 2013/14.

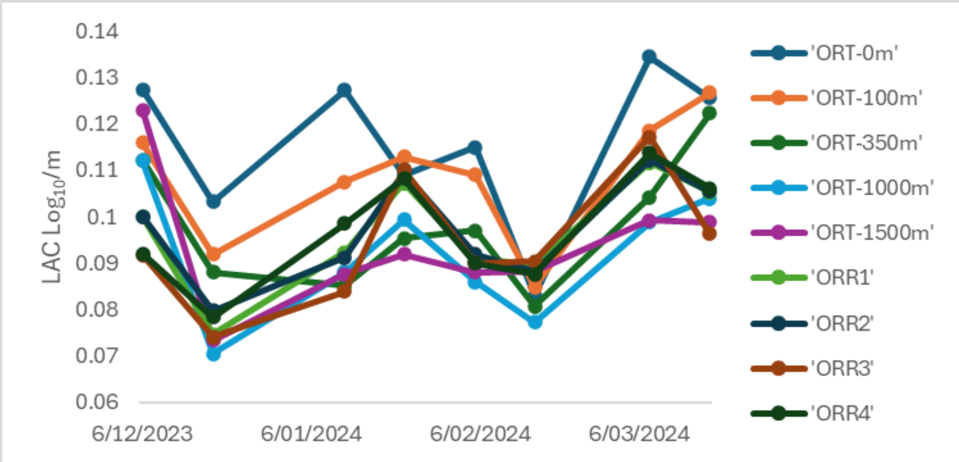


Figure 25: Light attenuation coefficient values recorded during monitoring across all sites undertaken between 6 December 2023 and 19 March 2024.

3.3.3.2. Nutrient enrichment

A composite sample representative of the top half of the water column is collected from each of the monitoring sites for analysis of chlorophyll-a and nutrients. Nutrient data provided by WaterCorp extend from 9 December 2022 to 19 March 2024 to capture data over the recent two summer periods. The results for ammonia (NH_4), nitrate/nitrite (NO_x), ortho-phosphate (Ortho-P) and chlorophyll-a (Chl-a) are provided in Figure 26. The results indicate increased nutrients in receiving waters in the 2023/24 summer compared to the previous year.

Ammonia was only detected above the laboratory limit of reporting (LOR) of <3 µg/L at the outfall site on three surveys in 2022/23: 9 and 17 January 2023 (4 µg/L/5 µg/L) and 20 February 2023 (4 µg/L). All other ammonia results at other sites were below the laboratory LOR. Five surveys recorded concentrations above the LOR in 2023/24 which were higher than the previous summer and across multiple impact sites. On the 6 and 19 December 2023, concentrations above the LOR were recorded at both the outfall (15 µg/L/6 µg/L) and at 100 m (6 µg/L/4 µg/L). On 12 January 2024, a concentration of 4 µg/L was recorded from the outfall site. On the 5 February, concentrations above the LOR were recorded at the outfall (14 µg/L), 100 m (17 µg/L) and 350 m (12 µg/L) locations. On 23 January outfall concentrations were below the LOR, but 100 m and 350 m sites recorded ammonia concentrations of 6 µg/L and 3 µg/L, respectively.

Nitrate/nitrite averaged 34.4 µg/L across sites from all surveys, ranging from below the LOR to 230 µg/L. The average across sites between summers was comparable, with 2022/23 recording 32.7 µg/L compared to 36.2 µg/L for 2023/24. Mean concentrations across surveys gradually reduced with distance from the outfall concentration of 102.1 µg/L, recording 76.0 µg/L at 100 m, 56.3 µg/L at 350 m, 20.7 µg/L at 1,000 m and 10.0 µg/L at 1,500 m, compared to reference site means ranging between

6.7 µg/L and 9.3 µg/L. Elevated concentrations at the outfall site above 100 µg/L were recorded during surveys in December 2022, February and December 2023, January and February 2024.

Concentrations recorded for ortho-phosphate averaged 15.7 µg/L ranging from 3 µg/L to 110 µg/L, with comparable means between years of 15.9 µg/L in 2022/23 and 15.4 µg/L in 2023/24. Mean concentrations across surveys gradually reduced with distance from the outfall concentration of 45.6 µg/L, recording 33.3 µg/L at 100 m, 22.4 µg/L at 350 m, 9.8 µg/L at 1,000 m and 7.1 µg/L at 1,500 m, compared to reference site means ranging between 5.5 µg/L and 6.0 µg/L. Elevated concentrations at the outfall site above 50 µg/L were recorded during surveys in December 2022, January, February and December 2023, January and February 2024.

Productivity indicators such as chlorophyll-a are recommended in EPA (2016) to develop EQG for nutrient enrichment rather than nutrients, as the latter may be naturally variable. The mean chlorophyll-a concentration across surveys and sites was 0.6 µg/L, ranging from 0.1 µg/L to 2.5 µg/L. The summer of 2023/24 recorded higher average concentrations (0.8 µg/L) than 2022/23 (0.4 µg/L). The nutrient trends of declining concentrations with distance from the outfall were not identified for chlorophyll-a, although a discrepancy was still apparent between impact and reference locations. For example, the overall average across impact sites was 0.7 µg/L compared to 0.5 µg/L at reference. These values were 0.5 µg/L versus 0.3 µg/L for 2022/23 and 0.9 µg/L versus 0.7 µg/L for 2023/24, respectively. Therefore, results indicate an increase in background chlorophyll-a concentrations combined with a consistent discrepancy of enrichment recorded within receiving waters.

Concentrations >1 µg/L have only been recorded during surveys from January to March 2024. The survey on 8 March recorded particularly high concentrations ranging from 1.3 µg/L to 2.5 µg/L. These surveys with elevated concentrations exhibit very little evidence of anthropogenic input showing comparable means between impact and reference sites. The survey undertaken on 16 February 2024, for which the transect is orientated towards Mullaloo Beach, suggests concentrations at reference sites were higher than impact with means of 0.9 µg/L and 0.5 µg/L, respectively. This finding suggests chlorophyll-a concentrations at the outfall site and in nearshore shallower waters off Mullaloo Beach were lower than recorded at sites approximately 3 km south in deeper waters.

The findings are supported in BMT (2024). The EQG for nutrient enrichment using chlorophyll-a has not been met for the last three years. This EQG compares the median of the current HEPA sites against the historical reference 80th percentile of 0.4 µg/L. The EQS states that the EQG should not be exceeded in two consecutive years, which has now not been met for the last two years. The 80th percentile of reference sites in 2022/23 was also 0.4 µg/L, while the same value in 2023/24 was 1.1 µg/L. The EQG has exceeded four additional occasions historically and the EQS once beyond the last three years, with EQG exceedances in 2010/11, 2013/14, 2015/16 and 2016/17 (BMT 2024).

Further, EQGs for phytoplankton biomass, also measured as chlorophyll-a, were exceeded in the 2023/24 summer. This describes that the chlorophyll-a median from at least one survey, or 25% of surveys for each impact site, exceeded three times the historical reference median concentration of 0.6 µg/L. In the 2023/24 summer the median across HEPA sites and within each site exceeded 0.6 µg/L on five separate surveys. This also represents the third consecutive year of EQG exceedances, resulting in an EQS also not being met for the second consecutive year. The EQGs have been routinely exceeded historically, although the EQSs have only exceeded within the last two years. BMT (2024) note concentrations would not have exceeded if three times the reference site median had been calculated for the corresponding 2023/24 period (1.65 µg/L), indicating the current exceedances are primarily attributable to an increase in natural background concentrations.

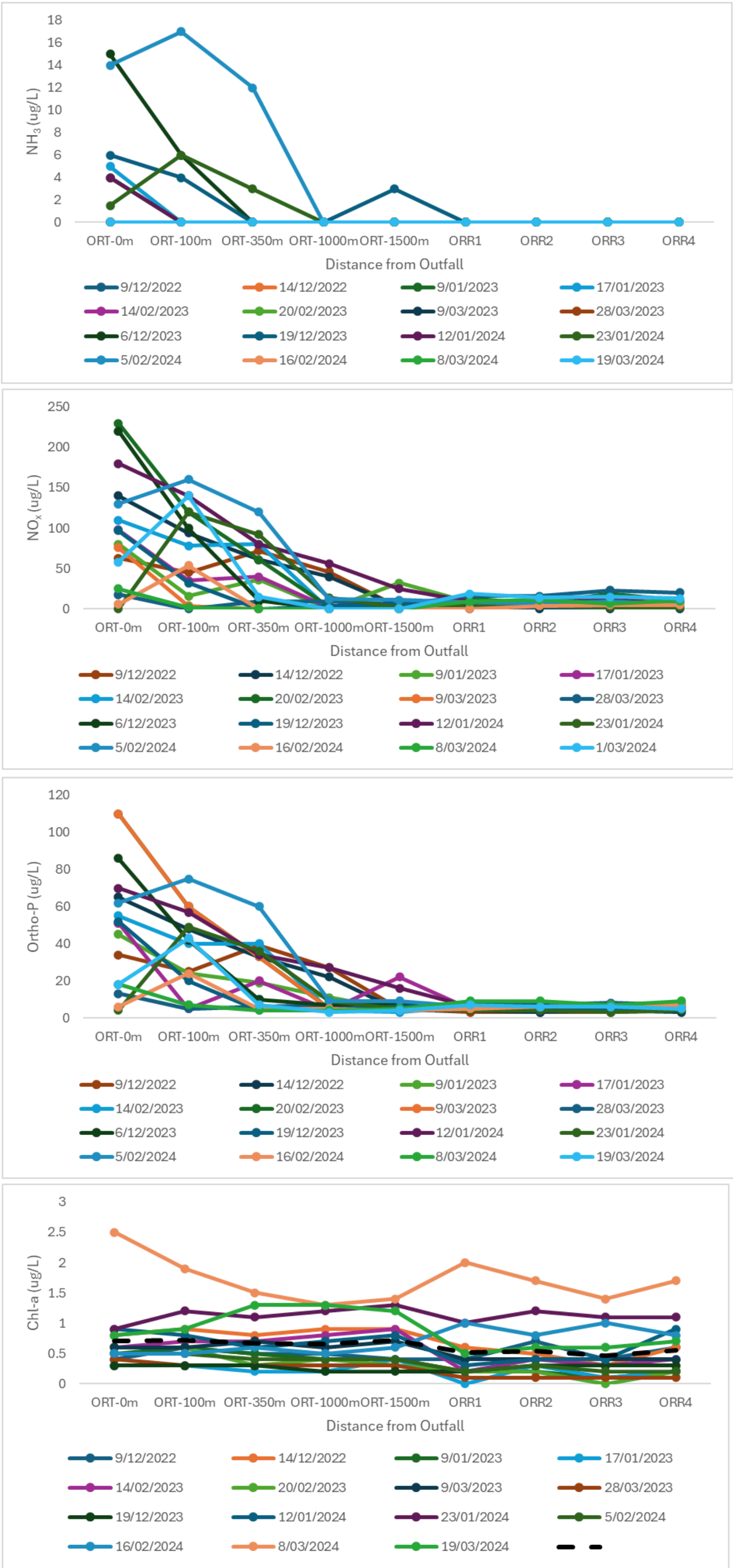


Figure 26: Nutrient (NH_4, NO_x, Ortho-P) and chlorophyll-a concentrations at recorded across all sites undertaken between 9 December 2022 and 19 March 2024.monitoring sites for 2022-2024

3.3.3.3. Pathogenic bacteria

Thermotolerant coliforms (TTC), *Enterococci* and *Escheria coli* are sampled and tested fortnightly during the summer period as part of the Beenyup Ocean Outfall Monitoring and Management Plan. This testing is undertaken to meet the EQO maintenance of seafood for human consumption as well as the EQO maintenance of primary and secondary recreation values. The seafood EQO determines whether there is potential for contamination of shellfish near the outlets with monitoring undertaken at the boundary of the observed zone of effect (OZE) and pre-designated Seafood Management Zone (Figure 27). The recreation EQO provides an indicator for disease-causing microorganisms (pathogens) associated with bathing areas, with monitoring undertaken at the boundary of the Primary Contact Management Zone (Figure 28). In total there are 31 Ocean Reef (OR) sampling locations, although coordinates were not provided. Five samples are also collected along gradient design sampling locations presented previously, as well as at eight shoreline monitoring locations (ORSL) for which locations are unknown.

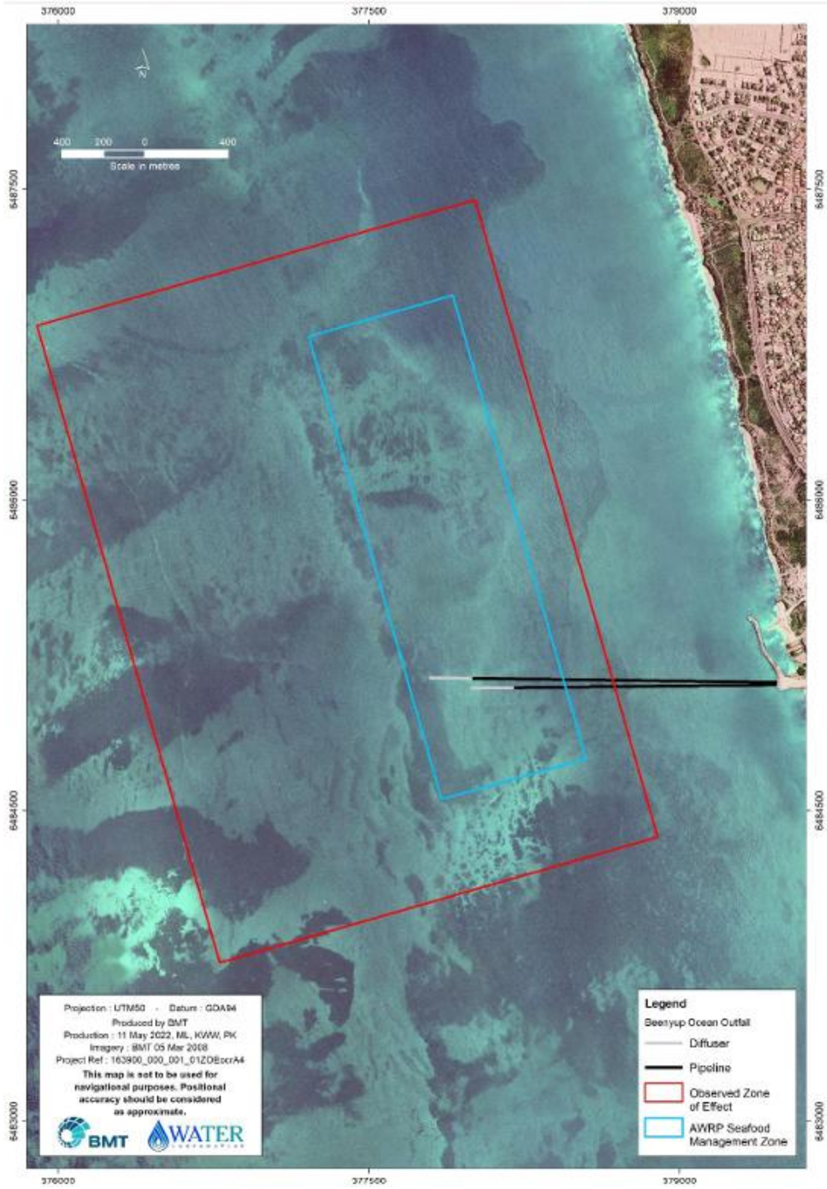


Figure 27: Boundaries of Observed Zone of Effect (OZE) and Seafood Management Zone around the Beenyup Ocean Outlet (BMT 2024)

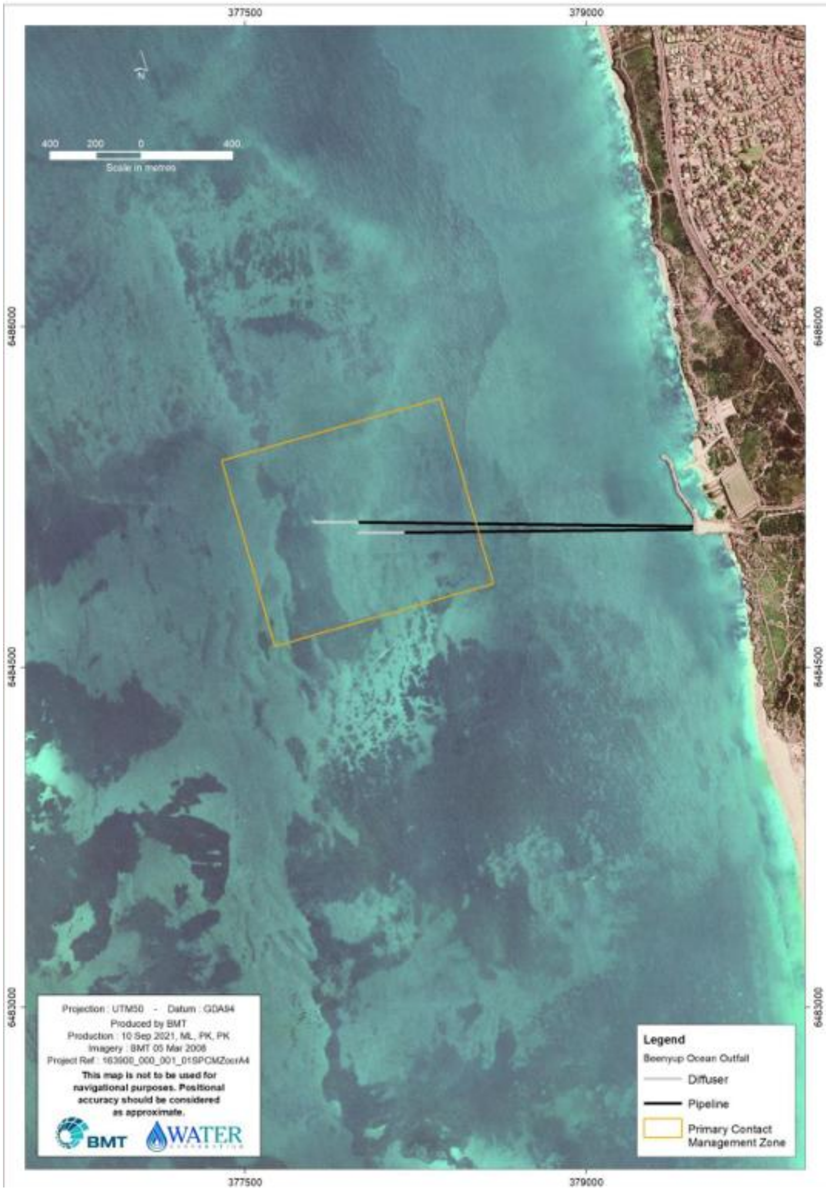


Figure 28: Boundaries of Primary Contact Management Zone around the Beenyup Ocean Outlet

The pathogenic sampling results recorded above the laboratory LOR from all site locations during the 2022/23 and 2023/24 summer sampling periods are presented in Table 12. The results indicate the risk of pathogenic bacteria is sufficiently diluted from concentrations in the TWW identified in Table 11. However, evidence of bacteria above typical concentrations was recorded at shoreline sites on 23 January 2024.

The OR monitoring sites recorded concentrations of TTC and *E.coli* at the detectable limit of 10 CFU/100 ml on 28 March 2023 at two sites (OR26, OR27) and at one site on 5 February 2024.

Pathogenic bacteria were detected only in samples as part of the gradient sampling program during surveys implemented on 14 and 20 February, 28 March and 19 December 2023. All three indicators were recorded on 14 February at sites 100 m and 350 m from the outfall at concentrations of 10 or 20 MPN/CFU/100 ml. TTCs and *E. coli* were measured as 20 CFU/100 ml at the outfall and 10 CFU/100 ml at 100 m on 20 February. On 28 March and 19 December, *Enterococci* recorded a concentration of 10 MPN/100 ml at the outfall location and at 100 m distance, respectively.

Table 12: Results for pathogenic bacteria samples collected between 9 December 2022 and 18 March 2024 which were recorded above the laboratory LOR

Date	Site	Thermotolerant coliforms (CFU/100ml)	*Escheria coli* (CFU/100ml)	*Enterococci* (MPN/100ml)
Ocean Reef Monitoring Sites				
28 Mar 2023	OR26new	10	10	-
	OR27new	10	10	-
5 Feb 2024	OR24	10	10	-
Gradient Design Monitoring Sites				
14 Feb 2023	ORT_100 m	20	20	10
	ORT_350 m	20	10	10
20 Feb 2023	ORT_0 m	20	20	20
	ORT_100 m	10	10	10
28 Mar 2023	ORT_0 m	-	-	10
19 Dec 2023	ORT_100 m	-	-	10
Shoreline Monitoring Sites				
17 Jan 2023	OSRL2	10	10	-
	OSRL3	10	10	-
20 Feb 2023	ORSL4	20	20	-
	ORSL5	20	10	-
23 Jan 2024	ORSL1	20	20	-
	ORSL2	-	-	170
	ORSL4	-	-	1,000
	ORSL5	-	-	31
	ORSL6	-	-	20
	ORSL7	-	-	41
18 Mar 2024	ORSL2	-	-	10

The shoreline monitoring locations recorded concentrations above the laboratory LOR during four survey events. Two locations (OSRL2, OSRL3) recorded TTC and *E. coli* concentrations at the detectable limit of 10 CFU/100 ml on 17 January 2023, while two more sites (OSRL4, OSRL5) logged concentrations of 10 to 20 CFU/100 ml on 20 February 2023. On the 23 January 2024, one site (ORSL1) recorded TTC and *E. coli* at 20 CFU/100 ml and a further five sites recorded *Enterococci* at higher concentrations ranging from 20-1,000 MPN/100 ml. The site OSRL2 also recorded *Enterococci* at 10 MPN/100 ml on 18 March 2024.

The pathogenic bacteria EQG for the seafood EQO has not been exceeded during historical sampling on the Beenyup Ocean Reef monitoring program (BMT 2024). This describes that the median TTC concentrations at sites at the boundary of the OZE have not exceeded 14 CFU/100 mL and the 90th percentile of TTC concentrations have not exceeded 21 CFU/100 mL. The EQG for primary contact have also not been exceeded, which means the 95th percentile bacterial content of marine waters has not exceeded 200 *Enterococci* MPN/100 mL. Shoreline sites are not formally assessed against the EQC, although the results from these sites did not exceed the TTC and *Enterococci* EQG for seafood and primary contact recreation applied to the OR monitoring sites above.

3.3.3.4. Phytoplankton

Phytoplankton samples are collected at the same locations on the boundary of the OZE and Seafood Management Zone, as well as the primary contact recreation boundary, shown in Figure 27 and Figure 28 where pathogenic bacteria are sampled. The seafood EQG for toxic phytoplankton species states that concentrations of potentially toxic algae are not to exceed the Western Australian Shellfish Quality Assurance Program (WASQAP) guideline concentrations in any samples (DoH 2020). The list of potentially toxic algae species and their cell count guideline for the EQG include any of the following:

- *Alexandrium* spp. (200 cells/L)
- *Dinophysis* spp. (1,000 cells/L)
- *Gymnodinium catenatum* (1,000 cells/L)
- *Karenia brevis* (1,000 cells/L)
- *Karenia/Karlodinium/Gymnodinium* group (250,000 cells/L)
- *Prorocentrum lima* (500 cells/L)
- *Pseudo-nitzchia* group (500,000 cells/L)

In addition to the above, environmental criteria have been derived for toxic algae in marine recreational water (DoH 2022). The EQG is described as:

A. The phytoplankton cell count* from a single site, should not:

   Exceed 10,000 cells/mL, or;

   Detect 'WA Health watch list' (Table 13) species or exceed their trigger levels.

B. There should be no reports of skin, eye or respiratory irritation or potential algal poisoning in swimmers considered by a medical practitioner as potentially resulting from toxic algae.

Exceedance of 'WA Health watch list' trigger levels, should action re-sampling of the site within 72 hours of identification of the exceedance for assessment against EQS and a visual assessment for algal scum within the defined recreational area.

Table 13: The 'WA Health watch list' for potentially toxic algae in marine recreational waters

Algal Group	Algal Genus /Complex	Key Species	Trigger Levels		Action Levels		
			Cell Counts cells/mL	Biovol. Mm/L3	Cell Counts cells/mL	Biovol. mm/L3	Visual Observation
Cyanobacteria	*Lyngbya*	*majuscula*	Detected				Algal filaments - widespread
	Trichodesmium	spp.	≥5,000	≥0.4	≥50,000	≥4	Algal scums
	Other	spp.		≥0.4		≥4-10	Algal scums
Dinoflagellates	*Alexandrium*	spp.	≥1*		≥10**		
	Karenia	*brevis*	≥5*		≥10**		
	Karenia	spp.	≥50*		≥100**		
	Pfiesteria	spp.	Detected				Algal scums
phytoplankton			≥10,000		≥50,000		

Review of historical sampling results indicate *Gymnodinium* and *Pseudo-nitzchia* taxa commonly occur in samples collected from sites in the area. *Dinophysis,* and *Prorocentrum* taxa have occasionally been recorded within survey sites, while *Alexandrium* taxa was recorded in 2018 and *Karlodinium* and *Karenia* taxa have been recorded in recent surveys since 2023 (BMT 2024).

There were no instances where toxic phytoplankton species were present at densities greater than the WASQAP (DoH, DPIRD and Industry, 2020) guideline values in 2023/24. Historically, the EQG has been exceeded on one occasion, which was recorded in 2019/20. This exceedance occurred on 27 March 2020, where the toxic phytoplankton species of the *Pseudo nitzschia seriata* group (101,520 cells/L) were recorded at greater density than 50,000 cells/L, the recommended WASQAP guideline value for this species. A concentration of 97,440 cells/L of the same species (i.e. above the guideline value) was also recorded at the reference site (ORR1), suggesting the phytoplankton species was widespread and not related to the operation of the outlet (BMT 2024).

The median total phytoplankton cell count from all the samples should not exceed 10,000 cells/mL or contain species on the Department of Health (DoH) watch list exceeding their trigger levels to meet the EQO for primary contact recreation. The algal density did not exceed, or has never exceeded, 10,000 cells/mL at any site. However, *Trichodesmium*, a DoH watch list species, exceeded trigger level concentrations for this species in 2022, but remained below 50,000 cells/ml when compared to the EQS.

3.3.3.5. Aesthetic observations

Routine monitoring for the Beenyup Ocean Outfall Monitoring and Management Plan includes assessment of the quality of surface water appearance during each survey. The protocol for the survey activity involves recording observations on nuisance organisms, fauna deaths, water clarity, colour or odour, and the presence of surface films or debris. The field surveys in 2023/24 found algae/plant material comprised of seagrass wrack visible on the surface on 50% of occasions and a noticeable green colour variation on 75% of occasions. The colour variation results exceeded the EQG and triggered comparison with the EQS, which is based on the community's perception of aesthetic value. Complaints were received by the community during the 2023/24 period regarding *Trichodesmium* blooms at Mullaloo Beach. BMT (2024) conclude these complaints are unrelated to the operation of the outfall so an EQS was not exceeded.

3.4. Microbial Testing Program

Swimming in polluted waters is a well-known risk to human health, and accordingly, the Western Australian Department of Health (WA Health) administers a microbial water monitoring program administered under the National Health and Medical Research Council (NHMRC). Guidelines for Managing Risks in Recreational Water (NHMRC Guidelines 2008) manage health risks from environmental (coastal, estuarine and freshwater) recreational water in Australia. EQG for faecal contamination for both primary (swimming snorkelling and diving) and secondary (boating and fishing) recreational contact with water bodies are based on the 95th percentile of *Enterococci* concentrations remaining below 200 MPN/100 mL and 2,000 MPN/100 mL, respectively.

Weekly sampling for faecal pathogens (*Enterococci spp*) was undertaken from 2 November 2022 to 21 April 2023, and again between 20 November 2023 and 27 March 2024, at three coastal sites: Key West, Mullaloo and Pinnaroo. During this time, most samples (86%) resulted in *Enterococci* being recorded at or below the laboratory LOR (<10 MPN/100 mL). Results with concentrations above the laboratory LOR are presented in Table 14.

As the minimum number of 100 samples was not met, the 95th percentile was not calculated (NHMRC 2008), although the interrogated dataset did not show cause for concern that the guideline for primary contact would be exceeded. There was only one exceedance of the primary contact concentration recorded at Pinnaroo on 2 November 2022 (640 MPN/100 mL). Elevated concentrations were recorded on 31 Jan 2024 at 100 MPN/100 ml, but did not exceed the EQG for primary contact. This corresponded with detectable concentrations recorded from the ocean outfall monitoring program at numerous beach locations on the 23 January 2023.

Six results were detected above the laboratory LOR during the 2022/23 season compared to only two in 2023/24. Mullaloo Beach and Key West detected *Enterococci* concentrations above the LOR on only one occasion. Comparably, Pinaroo Point detected *Enterococci* concentrations on seven occasions.

Table 14: Department of Health sampling results with concentrations above the laboratory LOR between 2 Nov 2022 and 27 Mar 2024

Date	Samples	Sample Location	Result
02 Nov 22	3	1x survey each at Key West, Mullaloo and Pinaroo Point	Mullaloo and Key West <10MPN/100mL Pinaroo *Enterococci* 640MPN/100mL
28 Feb 23	3	1x survey each at Key West, Mullaloo and Pinaroo Point	Pinaroo *Enterococci* 40MPN/100mL Mullaloo *Enterococci* 20MPN/100mL Key West <10MPN/100mL
15 Mar 23	3	1x survey each at Key West, Mullaloo and Pinaroo Point	Mullaloo and Key West <10MPN/100mL Pinaroo *Enterococci* 41MPN/100mL
22 Mar 23	3	1x survey each at Key West, Mullaloo and Pinaroo Point	Mullaloo and Key West <10MPN/100mL Pinaroo *Enterococci* 20MPN/100mL
27 Mar 23	3	1x survey each at Key West, Mullaloo and Pinaroo Point	Mullaloo and Key West <10MPN/100mL Pinaroo *Enterococci* 20MPN/100mL
21 Apr 23	3	1x survey each at Key West, Mullaloo and Pinaroo Point	Mullaloo and Pinaroo <10MPN/100mL Key West *Enterococci* 40MPN/100mL
31 Jan 24	3	1x survey each at Key West, Mullaloo and Pinaroo Point	Mullaloo and Key West <10MPN/100mL Pinaroo *Enterococci* 100MPN/100 mL
28 Feb 24	3	1x survey each at Key West, Mullaloo and Pinaroo Point	Mullaloo and Key West <10MPN/100mL Pinaroo *Enterococci* 20MPN/100 mL

Historical monitoring for bacteria testing presented in Strategen (2016) and sourced from DoH (2016) was undertaken at six locations; Burns Beach, Ocean Reef Boat Harbour Entrance, Ocean Reef Boat Harbour (1st platform near toilets), Key West, Mullaloo Beach and Pinnaroo Point. An assessment of 'Sanitary Inspection' graded Burns Beach, Ocean Reef Harbour (Beach), Mullaloo Beach and Pinnaroo Point as only "Fair". This grade describes that bacterial water quality results are typically 'Good', although animal pollutant sources (e.g. bird faeces) may elevate bacterial levels following rainfall. The remaining sites were classified as "Good". Therefore, detectable concentrations of *Enterococci* during 2022/23 and 2023/24 sampling were likely primarily attributable to animal pollutant sources.

3.5. DWER Environmental Response

As part of the response to managing marine water quality in WA, DWER conducted five surveys following reported poor water quality at Mullaloo beach between 17 January – 26 April 2024.

A total of 42 water quality samples were collected. The sampling locations, analytes and a summary of the algal findings are presented in Table 15. Samples were generally collected from Mullaloo Beach and at the southern rockwall of the Ocean Reef Marina. A comprehensive sampling program was undertaken on 25 March 2024, with 10 sites each monitored on the shoreline and 200-300 offshore at 500 m intervals from Pinaroo Point to Ocean Reef Marina, as well as six sites in the vicinity of the Beenyup outfall and two background sites approximately 2 km north of Ocean Reef Marina at 80 m and 300 m offshore (Figure 29). Water samples were typically laboratory tested for algae identification by the DWER Phytoplankton Ecology Unit (PEU). Additional testing was undertaken by external laboratories on samples collected on 21 and 25 March, and 24 and 26 April 2042, which included the following constituents:

- TRH (C6-C40) / BTEXN / PAH / Metals (Total & Dissolved)
- Nitrite, Nitrate, Ammonia, Reactive Phosphorus, Total Nitrogen, TKN, Total, Phosphorus
- Ionic Balance:
    - Anions: Major (Cl, SO4, Alkalinity), Fluoride
    - Cations: Major (Ca, Mg, Na, K), Hardness
- Algal biomass

3.5.1. Toxicants

Total recoverable hydrocarbons (TRH), benzene, toluene, ethyl benzene, xylene and naphthalene (BTEXN) and polyaromatic hydrocarbons (PAHs) were recorded below the laboratory LOR and were not detected in all samples collected during DWER monitoring in 2024. Total and dissolved metals also mostly recorded concentrations below the LOR. The exceptions included consistently low natural concentrations of Boron and Barium, detection of low total chromium and manganese at S3 on 25 March, but dissolved metal results for these constituents remained below the LOR, and one result for zinc at site MB2 on 21 March which exceeded the 80% species protection level in both total and dissolved concentrations (Table 16). This elevated zinc result likely indicates some contamination of the sample from sunscreen or other external factors while collecting the sample in the field. Ammonia concentrations remained below the ANZG (2018) default guideline value of 500 µg/L.

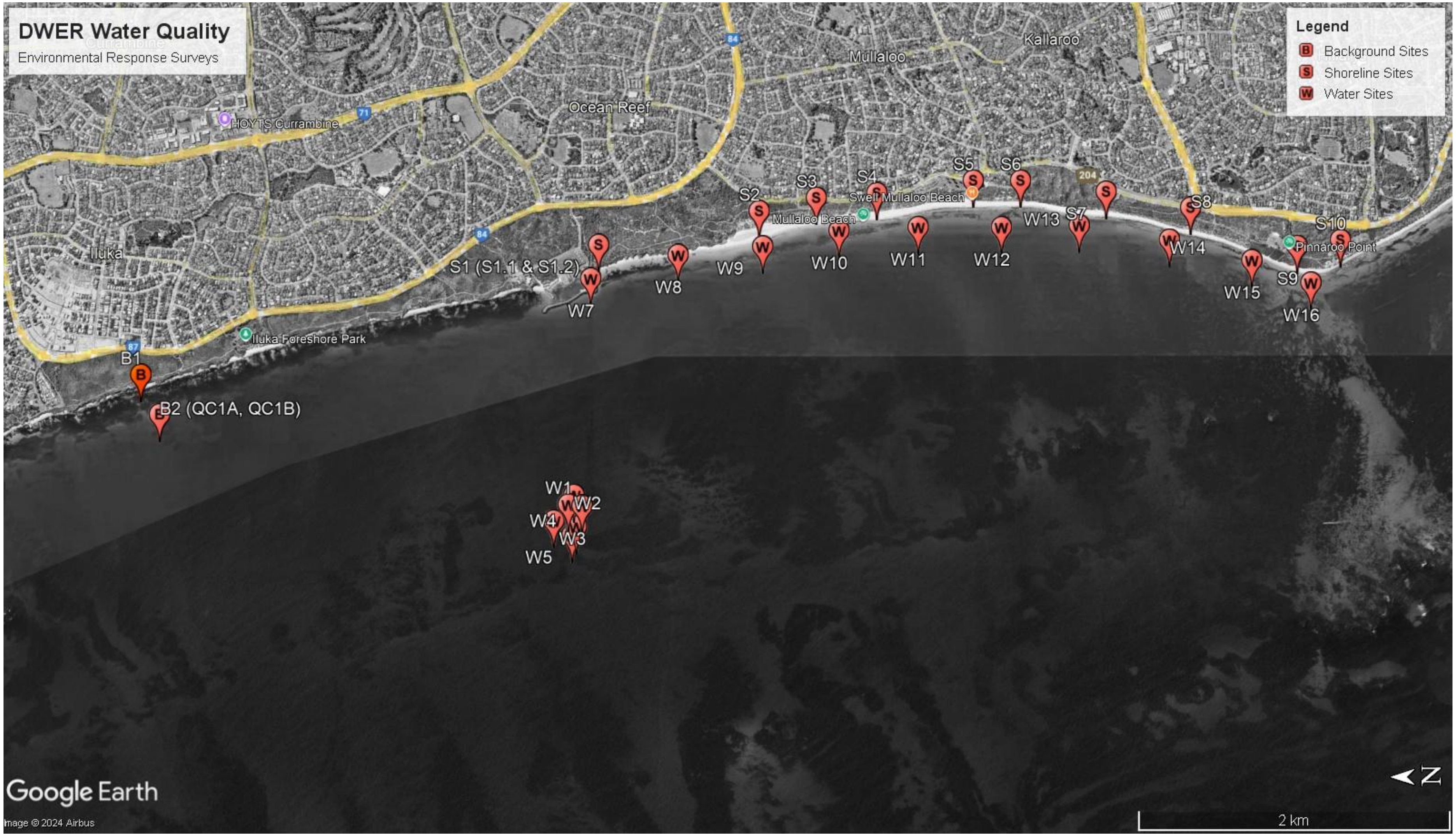


Figure 29: DWER water quality sites surveyed during the Environmental Response Surveys at Mullaloo Beach between 17 January – 26 April 2024. Water sites (W), Shoreline sites (S) and Background sites (B).

Table 15: DWER environmental response sampling summary between 17 Jan and 26 Apr 2024

Date	Owner	Samples	Sample Location	Analyte	Result
17/1/2024	DWER	2	Ocean Reef Marina Southern Rockwall	Algae	*Trichodesmium*
02/2/2024	DWER	2	Ocean Reef Marina (groin) Mullaloo Beach	Algae	No potentially harmful phytoplankton species were observed at levels of concern.
26/2/2024	DWER	2	Mullaloo Beach south of surf club Ocean Reef Marina southern rockwall	Algae	No potentially harmful phytoplankton species were observed at levels of concern.
21/3/2024	DWER	2	Mullaloo Beach south of surf club Mullaloo Beach north of surf club	Algae and physio chemical parameters, Metals, Nutrients, Anions, Cations.	No potentially harmful phytoplankton species were observed at levels of concern.
22/3/2024	DWER	1	Mullaloo Beach (In between Flags)	Algae	No potentially harmful phytoplankton species were observed at levels of concern.
25/3/2024	DWER	10 10 6 2	Mullaloo Shoreline sites Mullaloo Ocean Water Sites Ocean Outfall sites Background sites	Algae and physio chemical parameters, Metals, Nutrients, SS, Bacteria, Anions. Algae ID for outfall sites only	Samples analysed for algae showed presence of potentially harmful phytoplankton. Thresholds for toxin producing algae not exceeded
24/4/2024	DWER	2	Mullaloo Beach North of Surf Club	Physio chemical parameters, Metals, Nutrients, SS, Bacteria, Anions.	*Trichodesmium* identified by PEU, broadcast provided.
26/4/2024	DWER	3	Mullaloo Beach North of Surf Club	Physio chemical parameters, Metals, Nutrients, SS, Bacteria, Anions.	*Trichodesmium* identified by PEU, broadcast provided.

3.5.2. Physico-chemical

The results recorded for physico-chemical parameters were typical of natural marine waters (Table 16). The pH remained relatively stable across sites sampled ranging from 8.03 to 8.24. Concentrations of salinity and total dissolved salts dropped in April, likely related to a decrease in water temperature at the end of summer, ranging from 35.2 psu to 37.6 psu and from 34,600 mg/L to 36,700 mg/L, respectively. Suspended solid concentrations (SSCs) in the water column were variable ranging from <5 mg/L to 171 mg/L. Highest SSCs were measured in samples collected from the shoreline in March and April. SSCs detected in these samples likely represents elevated microalgal and marine debris causing the discoloured water appearance reported from the public. The marine waters were expectedly high in natural minerals, in particular calcium and magnesium salts, represented by concentrations ranging between 5,160 mg/L and 7,230 mg/L. Alkalinity concentrations were also in the high range for marine waters helping to regulate alkaline pH values, ranging between 126 mg/L and 134 mg/L.

3.5.3. Nutrient enrichment

Measured concentrations of nutrients during DWER sampling were variable and at times elevated. TN concentrations ranged from below the laboratory LOR, measured as either 100 or 200 µg/L, to 700 µg/L. Highest measured concentrations were recorded from samples collected at the shoreline on 21 March and 24/26 April. Ammonia (NH_4) ranged from 5 µg/L to 210 µg/L, with the highest concentrations typically measured at Ocean Reef Marina sites on 25 March and elevated concentrations recorded in April. Nitrate/nitrite (NO_x) concentrations range from 5 µg/L to 170 µg/L, with highest concentrations recorded around the outfall on 25 March. These concentrations generally reflect those measured at the outfall and within the LEPA at 100 m from the outfall (see Section 3.3.2).

The results indicate highest TN concentrations are associated with organic nitrogen and likely depict the presence of a blue-green algae *Trichodesmium* bloom. These algae are 'nitrogen fixers' making inorganic nitrogen bioavailable, which is represented by spikes in ammonia first then nitrate/nitrite following bloom detection. The algae do not survive for long in these surface concentrations for various reasons and start to decompose which creates additional bioavailable nitrogen for uptake by plants. Outfall monitoring indicates nitrogen is released in both organic and inorganic forms, with inorganic concentrations primarily represented in nitrate/nitrite. However, review of gradient monitoring data indicates effects are rarely detectable in any minor increase from background at 1,500 m from the outfall in the direction of current.

TP concentrations ranged from <20 µg/L and 870 µg/L. TP was high on 21 March, which likely indicates the abundance of reported algae and marine debris tissue in organic form. This is evidenced by low ortho-phosphorus concentrations (4-5 µg/L). Ortho-phosphorus ranged from below the LOR to 60 µg/L. Results indicate the highest ortho-phosphorus concentrations were recorded at sites around the outfall on 25 March. Like nitrogen, these concentrations generally reflect those measured at the outfall and within the LEPA at 100 m from the outfall which are rarely detectable in any minor increase from background at 1,500 m from the outfall in the direction of current (see Section 3.3.2).

Table 16: Summary of physico-chemical, zinc and nutrient results from DWER environmental response sampling summary between 17 Jan and 26 Apr 2024

Parameter	Site/ Statstic	Physico-chemical						Dissolved metals	Nutrients				
		pH	Salinity (psu)	TDS	SSC	Hardness (CaCO3)	Total alkalinity (CaCO3)	Zinc	NH3	NOx	TN	TP	Ortho-P
EQC		-	-	-	-	-	-	3.3	500	-	-	-	-
21-Mar	MS1					5,550	132	<25.0	5	5	660	870	4
	MB2					5,160	133	**28**	7	14	580	850	5
25-Mar S1-S10	Mean	8.13	36.42	35740	59.1	6503	133.8	<25.0	66	32.5	362.5	107	<10
	Min	8.08	36.2	35600	6	6380	132	<25.1	50	5	200	30	<10
	Max	8.15	36.7	36000	171	6600	136	<25.2	110	60	500	310	20
25-Mar W1-W6	Mean	8.14	36.58333333	35,883	18.6	6,498	128.5	<25.3	56.7	110	300	75	41.7
	Min	8.03	36.4	35,700	8	6,360	128	<25.4	30	30	300	60	20
	Max	8.17	36.8	36,100	26	6,680	129	<25.5	80	170	300	90	60
25-Mar W7-W16	Mean	8.16	36.85	36080	8.6	6485	131.6	<25.0	91	15	<200	38	9.5
	Min	8.14	36.5	35800	7	6390	130	<25.0	30	20	0	20	10
	Max	8.18	37.6	36700	18	6560	133	<25.0	210	30	0	60	20
25-Mar	B1	8.24	36.6	35900	<5	6470	130	<25.0	40	<10	<200	<20	<10
	B2	8.14	35.8	35200	<5	7230	128	<25.0	<50	<10	<100	20	<10

Parameter	Site/ Statstic	Physico-chemical						Dissolved metals	Nutrients				
		pH	Salinity (psu)	TDS	SSC	Hardness (CaCO3)	Total alkalinity (CaCO3)	Zinc	NH3	NOx	TN	TP	Ortho-P
24-Apr	W1	8.08	35.2	34,700	14	6,540	129	<25.0	130	90	500	40	<10
	W2	8.06	35.2	34,600	71	6,380	129	<25.0	100	90	700	90	<10
26-Apr	W1	8.03	35.8	35,200	33	6,410	126	<25.0	180	20	400	80	<10
	W2	8.07	36	35,400	17	6,530	128	<25.0	70	20	300	40	<10
	W3	8.07	35.8	35,200	7	6,500	127	<25.0	40	30	200	40	<10

3.5.4. Cations and Anions

Concentrations of cations and anions in marine water are higher in comparison to TWW (see Section 3.3.2.4) due to the increase in dissolved solids. Anions are dominated by chloride while cations primarily comprise sodium, followed by magnesium, calcium and potassium. The ionic balance results ranged between 0.04% and 7.97%, which is within the acceptable range of 10%.

3.5.5. Pathogenic Bacteria

Microbial testing was undertaken on samples collected no 25 March which included both *E. coli* and *Enterococci* indicators. Results typically reported concentrations below the LOR. Highest concentrations were recorded from shoreline samples, with *E. coli* ranging between <1 CFU/100 ml and 15 CFU/100 ml, while *Enterococci* ranged from <1 CFU/100 ml to 39 CFU/100 ml. Higher concentrations were recorded to the south of Mullaloo Beach near Pinaroo Point. These findings are consistent with the DoH microbial testing program where minor concentrations were detected near Pinaroo Point likely primarily attributable to animal pollutant sources. Results for water samples collected from the outfall, along Mullaloo Beach and at background sites ranged between <1 CFU/100 ml and 2 CFU/100 ml.

3.5.6. Phytoplankton

Samples tested by PEU on 17 January identified the presence of the filamentous Cyanobacteria *Trichodesmium cf erythraeum* on the 'WA Health watch list' (Table 13). Findings suggest the aggregation of *Trichodesmium* is likely to be the cause of the brown surface scum at Mullaloo Beach described in complaint observations submitted to Pollution Watch. *Trichodesmium*, more commonly known as sea sawdust, is ubiquitous in global tropical and subtropical oceans and is particularly abundant around Australia. In most cases the blooms are harmless, although if allowed to stagnate, *Trichodesmium* can release a clear, water-soluble toxin. Generally, the concentration of this toxin in natural systems is not sufficiently high enough to pose a threat to human health. However, it does have the potential to cause skin irritation or other allergic reactions.

PEU did not identify potential harmful phytoplankton species at levels of concern within samples collected from Mullaloo Beach and the Ocean Reef Marina on 2 and 26 February, and 21 and 22 March. Visible brownish 'tufts' in the water column were described, depicting marine debris likely comprised of clumped seagrass, macrophytes, rotifers, ciliates, zooplankton, empty diatom frustules, live diatoms, broken shells and fish scales. These were clearly distinct in structure and much larger than phytoplankton within the samples (Figure 30). Common diatoms from the genera *Leptocylindrus* spp., *Proboscia* spp. and *Asterionellopsis* spp. composed the highest densities within the samples, which at the densities observed could possibly cause slight discolouration in water, although not at levels the public would normally find aesthetically concerning. Therefore, the discolouration complaints were considered likely due to the aggregation of marine debris.

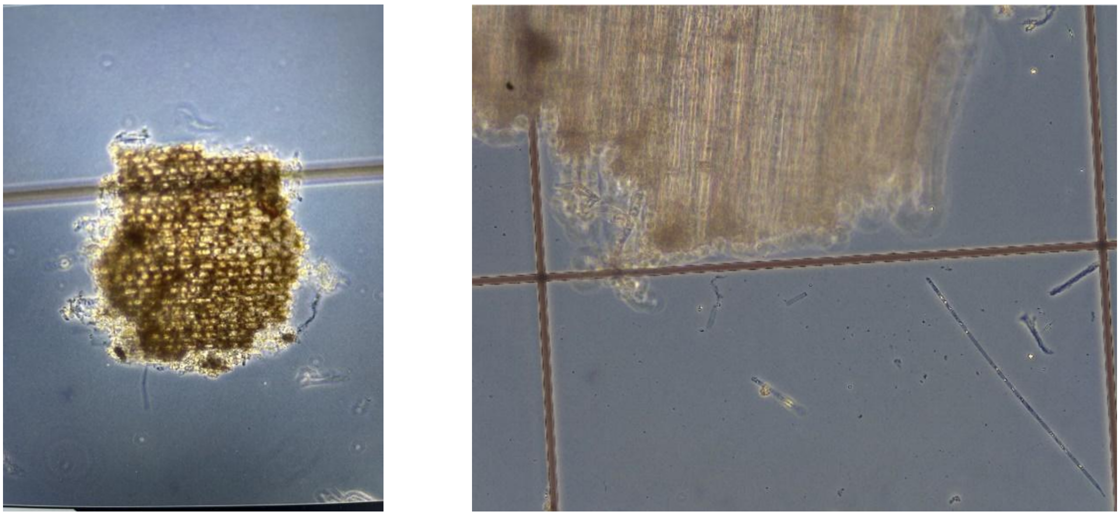


Figure 30: Microscopic comparison of the 'tufts' of marine debris (left) to the large (1 mm average) diatom, *Proboscia alata* which was common in the samples from 21 and 22 March 2024

Water samples collected on 25 March were assessed for phytoplankton identification and enumeration by two external NATA accredited laboratories. A summary of the algal sampling results are presented in Table 17. Similar to earlier March samples assessed by PEU, the phytoplankton assemblage at all sites was dominated by diatoms, particularly *Leptocylindrus* spp., *Cylidrotheca closterium*, *Proboscia alata* and *Pseudo-nitzschia* "*seriata*" group. The laboratory described that the total cell densities were high but not unusual, and results are typical when comparing against samples observed in coastal Perth waters for the previous month or so and the time of year. These concentrations are unlikely to have been high enough to be the cause of any water discolouration.

The highest mean density of total algae was recorded from the offshore Mullaloo Beach area, followed by the Beenyup Ocean Outfall, shoreline Mullaloo Beach. The density of potentially toxic algae was comparable across these areas. The reference locations 2 km to the north recorded significantly lower concentrations of both total and potentially toxic algae.

Potentially toxic species found in samples typically relate to the WASQAP guidelines for seafood (i.e., it relates to the consumption of shellfish) rather the EQO for primary contact. Exceedances of the WASQAP (2020) Alert Level of 50,000 cells/L for *Pseudo-nitzschia* spp. was recorded across all samples tested at one of the laboratories. It is noted results from this laboratory did not distinguish the group strand for this genus and recorded concentrations in cells/ml, requiring multiplication of results by 1,000 to convert the units for potentially toxic algae. The Alert Level of 50,000 cells/L is only applicable to the *Pseudo-nitzschia* '*seriata*' group. However, the Alert Level of 500,000 cells/L for the *Pseudo-nitzschia* '*delicatissima*' group was exceeded for the shoreline sample S4 analaysed by the same laboratory. The second laboratory confirmed phytoplankton levels above The Alert Level of 50,000 cells/L for *Pseudo-nitzschia* '*seriata*' group at the outfall sample W6, although this is located inside the OZE and Seafood Management Zone around the Beenyup Ocean Outlet. Exceedance of the WASQAP guideline values initiate flesh testing in mollusc species, but this should not be an issue for overall water quality or pose a primary contact risk.

Sampling undertaken again on 24 and 26 of April 2024 from Mullaloo Beach identified the presence of the filamentous Cyanobacteria *Trichodesmium*. The PEU issued warning notifications through the Department of Health and other government websites avoid primary contact with discoloured water or scum is visible, as it may lead to skin irritation, ear infections, burning throat or gastro-intestinal illness.

Table 17: Summary of total algae and total potentially toxic algae recorded from sites sampled on 25 March 2024

Sites/ Statstic	Total Algae			Potentially Toxic Algae			Taxa
	Mean	Min	Max	Mean	Min	Max	
S1-S10	1,260	610	2,100	228	9	675	*Pseudo-nitzschia* "*delicatissima*" group *Pseudo-nitzschia* "*seriata*" group *Oscillatoria* spp *Prorocentrum rhathymum*
W1-W6	1,439	476	3,180	208	17	619	*Pseudo-nitzschia* "*delicatissima*" group *Pseudo-nitzschia* "*seriata*" group GK Complex (*Gymnodinium-Karenia* Complex)
W7-W16	1,649	871	2,840	195	28	474	*Pseudo-nitzschia* "*delicatissima*" group *Pseudo-nitzschia* "*seriata*" group *Oscillatoria* spp *Phalochroma rotundatum* complex
B1	285	-	-	6	-	-	*Pseudo-nitzschia* "*delicatissima*" group *Pseudo-nitzschia* "*seriata*" group
B2	867	-	-	162	-	-	

3.6. Changing Trends in Water Quality

3.6.1. Treated Wastewater

Review of the data provided by DWER to characterise the TWW discharge identified there has been minimal detectable changes in discharge volumes and composition of constituents being measured in the effluent that could be considered a cause of concern.

Mean discharge volumes in 2024 are within the range of mean volumes discharged over the recent four years. No new toxicants have been detected within the TWW, with ammonia, copper and zinc the only toxicants historically requiring dilution within the LEPA to meet species protection levels. The concentrations of these toxicants were higher in 2024 than has been recorded historically since 2018. The WET testing also indicates the toxicity of the TWW has increased slightly since 2019, with a minimum concentration of 12.5% effluent in marine water for no effects. However, given worst-case estimated dilutions within the LEPA ranged between 1:134 and 1:385, typical dilutions achieved at the

LEPA boundary will be far greater than the minimum number of dilutions required for the effluent to not present a toxic risk beyond the LEPA boundary. For example, concentrations for ammonia, copper and zinc calculated at the LEPA boundary using estimated dilutions are well below ANZG (2018) guideline values, while WET testing indicates a minimum of only 1:8 dilutions will be required to achieve 12.5% effluent concentration.

Conductivity/TDS, SSC and BOD/COD concentrations within the TWW samples in 2024 were slightly higher than 2023. Higher salinity is not considered a concern as this will promote mixing with marine water (i.e. reduce potential for stratification). The mean SSC increased from 23.5 mg/L in 2023 to 39.2 mg/L in 2024, which suggests slightly higher concentrations of organic particulates in the discharge. This is supported by BOD results representing an increase in the amount of dissolved oxygen is required by aerobic biological organisms in 2023/24 to break down organic material. This is further supported by typically higher nutrients in the TWW for 2024 compared to 2023, although concentrations were slightly reduced from that recorded in 2022. Conversely, mean concentrations of pathogenic bacteria *E. coli* in 2024 was slightly lower than recorded in 2023.

3.6.2. Water Monitoring

Toxicants sampled for DWER monitoring showed no results indicative of contamination concerns. Physico-chemical data reviewed generally present relatively consistent values across the region, including impact and reference monitoring locations, typical of natural marine conditions. This is supported by very few historical exceedances of the EQG for salinity and dissolved oxygen in the Beenyup Ocean Outfall monitoring program (BMT 2024).

Salinity concentrations at the outfall location may occasionally be reduced due to the freshwater discharge which was no longer detectable at 100 m. The water temperature increased during measurements along a gradient transect oriented towards Mullaloo Beach on 16 February 2024, which is an atypical direction for currents during summer. These temperatures were comparably higher than reference sites further offshore. Dissolved oxygen recorded slightly elevated concentrations surrounding the outfall due to TWW discharge compared to the reference location, although may also be attributable to natural variation in dissolved oxygen between physical influences.

The SSCs measured during DWER sampling were variable, with elevated SSCs at the shoreline likely indicative of microalgal and marine debris identified in samples causing the discoloured water appearance reported from the public. Similarly, highest measurements of TN were recorded from shoreline sites during periods following *Trichodesmium* bloom detection in samples, with highest ammonia recorded at Ocean Reef Marina sites during subsequent surveys. Results imply the *Trichodesmium* bloom has triggered nitrogen fixation and nitrification, where dinitrogen is converted to ammonia, then ammonia is converted to nitrate/nitrite for promoting further algal growth.

Nutrient sampling in receiving waters for Beenyup Ocean Outfall indicates concentrations from the 2023/24 summer were slightly higher than compared to the previous year. The gradient monitoring design indicates the effects of elevated nitrogen and phosphorus at the outfall are gradually reduced and are barely detectable at 1,500 m in the direction of the current, indicating this site is only occasionally above background. Highest nitrate/nitrite and ortho phosphorus concentrations were also recorded around the outfall on 25 March in DWER sampling.

Results for chlorophyll-a suggest natural background concentrations have increased over recent years. This is supported by BMT (2024) recording consistent exceedances of historical background concentrations from reference sites, while comparison against reference site data for the corresponding periods would not have resulted in an exceedance. Chlorophyll-a means from impact sites over the last couple of years were still slightly higher than reference sites, although in 2024, surveys where elevated chlorophyll-a concentrations were recorded there was very little evidence of anthropogenic input. This indicates that when background concentrations are high, influence from the outfall is difficult to distinguish. Similar effects were observed for LAC, where effects are typically observed at the outfall, but concentrations can be lower than reference when natural background concentrations are high.

Occasional minor detections of *Enterococci* were recorded, particularly near Pinaroo Point, in all provided monitoring data, likely primarily attributable to animal pollutant sources. Evidence of elevated *Enterococci* above typical concentrations was recorded as part of outfall monitoring at shoreline sites on 23 January 2024. Concentrations from five shoreline locations ranged from 20 to 1,000 MPN/100 mL, with the highest concentrations exceeding the primary contact concentration (although not intended for instantaneous site comparison). The only other exceedance of the primary contact concentration from the data provided occurred at Pinnaroo on 2 November 2022 (640 MPN/100 mL). These results imply a more localised source of faecal material near the shoreline may have been responsible for detections on 23 January 2024.

Toxic phytoplankton species listed under the WASQAP have been historically recorded as commonly occurring within coastal waters. On 27 March 2020, toxic phytoplankton species of the *Pseudo nitzschia seriata* group were recorded at greater density than 50,000 cells/L, which is the recommended WASQAP guideline value for this species, at both an impact and reference site, indicating broadscale background elevated densities (BMT 2024). The DWER monitoring program identified a confirmed a density Alert Level of 50,000 cells/L for *Pseudo-nitzschia* '*seriata*' group at the outfall sample W6, although this is located inside the OZE and Seafood Management Zone around the Beenyup Ocean Outlet. The Alert Level of 500,000 cells/L for the *Pseudo-nitzschia* '*delicatissima*' group was exceeded for the shoreline sample S4 on 25 March 2024.

Phytoplankton samples collected from Mullaloo Beach on 17 January, and 24 and 26 of April 2024, identified the presence of the filamentous Cyanobacteria *Trichodesmium* on the 'WA Health watch list' for primary contact recreation. The density was not provided in data provided from DWER to enable comparison to DoH Watch List trigger levels for primary contact recreation. Findings suggest the aggregation of *Trichodesmium* is likely to be the cause of the brown surface scum at Mullaloo Beach described in complaint observations submitted to Pollution Watch. *Trichodesmium* has been reported during Beenyup Ocean Outfall monitoring previously in 2021/22, which exceeded the EQG trigger level concentrations of 5,000 cells/ml but remained below 50,000 cells/ml when compared to the EQS. Samples collected between 2 February and 22 March identified aggregations of visible brownish 'tufts' depicting marine debris (i.e. seagrass, macrophytes etc..) suspected to cause the discolouration complaints.

3.7. Influences on Water Quality

A variety of water quality datasets were analysed to investigate the potential effect of the Beenyup discharge on the water quality at Mullaloo beach. Ongoing data collection from the Beenyup ocean outfall, combined with environmental response and recreational use water quality monitoring, has enabled the assessment. Although individual exceedances of EQG's have been noted, and some noteworthy observations highlighted, a clear and consistent explanation of Beenyup ocean outfall contribution to recent marine water quality issues cannot be provided, due to limitations in the available data for evaluation.

Water quality is influenced by several natural and anthropogenic (human) factors. Natural influences are complex and variable (both seasonally and between years) and include currents, temperature, wave action, groundwater inputs and biological processes. Anthropogenic influences also have the potential to exacerbate or diminish natural influences, such as increasing nutrient loads in groundwater discharges to the ocean which adds a further degree of complexity to the system.

The water quality complaints received from the public appear to align with the occurrence of the 'WA Health watch list' Cyanobacteria *Trichodesmium* within samples collected in January and April 2024. *Trichodesmium* blooms are often mistaken for slimy effluent slicks on beaches when winds and tidal movements have blown the algae into coastal areas. Blooms are typically a rusty-brown colour, however some variations in colour may occur. In most cases the blooms are harmless, but if allowed to stagnate, *Trichodesmium* can release a clear, water-soluble toxin. Contact with these blooms can result in skin irritations such as stinging, tingling or a rash, particularly in people with sensitive skin. Other symptoms may include a sore throat, nausea and general weakness. It also smells offensive, like rotting plant matter or sulfur gas.

Trichodesmium are Cyanobacteria (blue-green algae) commonly known as sea sawdust that appear naturally in tropical and subtropical ocean waters but are particularly common around Australia. These algae are 'nitrogen fixers', which means that they can take nitrogen gas from air and 'fix' it in a form that can then be transferred into the food chain. Therefore, unlike other algal blooms, *Trichodesmium* are not necessarily produced from increase nutrient loads in marine waters. Qi et al. (2023) recently quantified the relative abundance of *Trichodesmium* around Australia which found surface aggregations almost everywhere except the southern coast. The spatial distribution and seasonality of occurrence were found to correlate well with temperature, sources of iron (Fe) to grow and fix nitrogen, as well as black carbon aerosols from frequent bushfires (Qi et al, 2023). Therefore, determining a link to the Beenyup ocean outfall may be challenging if the site represents conditions that are naturally conducive for this species, noting monitoring already reported a *Trichodesmium* bloom in the area two years prior. Other studies suggest phosphorus also plays a key role in *Trichodesmium* growth and fixation (Chinenye, 2023).

The Beenyup ocean outfall discharges TWW with elevated nutrients. However, monitoring of the discharge has demonstrated elevated concentrations are gradually reduced and are barely detectable above background concentrations at 1,500 m in the direction of the current. Analysis of current measurements near the Beenyup outlets suggest south-west currents towards Mullaloo Beach are uncommon in summer (Figure 13, Figure 14 and Figure 15). Therefore, nutrient input from the Beenyup outfall to Mullaloo Beach during atypical metocean conditions is likely to be minimal. Monitoring of the TWW suggests a slight increase in organic particulates while nutrient data indicates a couple of spikes in TN and TP between January and March 2024, although these changes were not detected during monitoring of receiving waters.

Of concern may be the increasing trend in background chlorophyll-a concentrations identified in recent annual monitoring reports for the Beenyup ocean outfall. Elevated chlorophyll-a has been recorded for three consecutive years, and in 2024 the median was twice the historical reference site 80th percentile value. These elevated concentrations have been recorded across both impact and reference sites indicating a regional change unrelated to the TWW discharge. Given there has been very little change to the TWW and discharge operations, other potential sources should be considered. The Ocean Reef Marina redevelopment and its potential alteration to localised ocean currents aligns with the timing of these observed changes, although changes in nutrient loads in groundwater, among other influences, could not be discounted.

Sampling evidence suggests Mullaloo Beach may have been influenced by nitrogen fixation and nitrification. While a *Trichodesmium* bloom fixes nitrogen, additional nutrients will become bioavailable and the decomposition of the algae will further enrich nutrients creating conditions favourable for algal growth. Measured concentrations of these nutrients were often elevated during sampling, particularly along the shoreline, but concentrations were isolated from that measured at the outfall. Warmer water temperatures were also recorded in shallower waters towards Mullaloo Beach. If coastal areas experience a natural *Trichodesmium* bloom, have warmer temperatures and reduced mixing from the marina, these factors may be conducive to eutrophication conditions.

Enterococci were also detected in elevated concentrations at shoreline sites on 23 January, indicating a local source of faecal matter near the shoreline may have been responsible for detections. Unconfirmed historical records indicate boats within the boat harbour present a likely source of contamination (Strategen 2016, DoH 2016). Poorly maintained sanitary waste systems aboard boats or poorly maintained pump-out stations at marinas can significantly increase bacteria and nutrient levels in the water. Consideration may therefore need to be afforded to nonpoint source pollution from the marina contributing to nutrient loads in coastal waters.

Targeted sampling and further assessment are needed to provide a definitive response to understanding the influence of the Beenyup ocean outfall on marine water quality. The natural, and potentially inevitable, occurrence of the water quality issues reported by the public should be considered, given the relative abundance of *Trichodesmium* blooms in Australia. Additionally, further studies may not be effective in identifying a solution to natural algal blooms in the future.

4. Near-Field Model of the Beenyup effluent

4.1. Overview

This section describes the near-field modelling of the Beenyup effluent, implemented to establish the optimal far-field modelling cell size (vertical and horizontal) in the vicinity of the discharge. The flow rates of the Beenyup outfall are presented, followed by the parameters considered in the near-field model. The results of the theoretical model are shown next, with the cell size used for the far-field model specified at the end of this chapter.

4.2. Beenyup Wastewater Treatment Plant Outfall

Outlet A terminates in a 200 m diffuser comprising 50 ports. Outlet B also terminates in a 200 m diffuser but comprises of 48 ports. Both diffusers have a pipeline diameter of 1.42 m, and discharge in approximately 10 m water depth (Water Corporation, 2023). Each port in the diffuser is a drilled hole with diameter of 0.16 m, oriented horizontally at the centre of the pipeline. Adjacent ports face opposite directions and are spaced approximately 4.05 m apart.

Beenyup outfall daily total discharge volumes since the year 2021, and flow rates discharged through each diffuser during the years 2023 to 2024, were provided by WaterCorp to DWER, who extended the information to O2Me for this study (refer Section 3.3). WaterCorp's discharge criteria for assigning flows through both pipes are unknown to O2Me, but this information is not critical to the context of this report. An analysis of average concurrent flow rates through each pipe revealed that the flows are almost always evenly distributed between both outlets (Figure 31). This simple finding is relevant to the setup of the numerical signal discharged in the hydrodynamic numerical model, as it will be shown in later sections of this report. Away from the discharge points, the proportion of total flow discharged through each diffuser is indistinguishable – both plumes merge and spread as one - therefore, it is reasonable to treat them as a single entity.

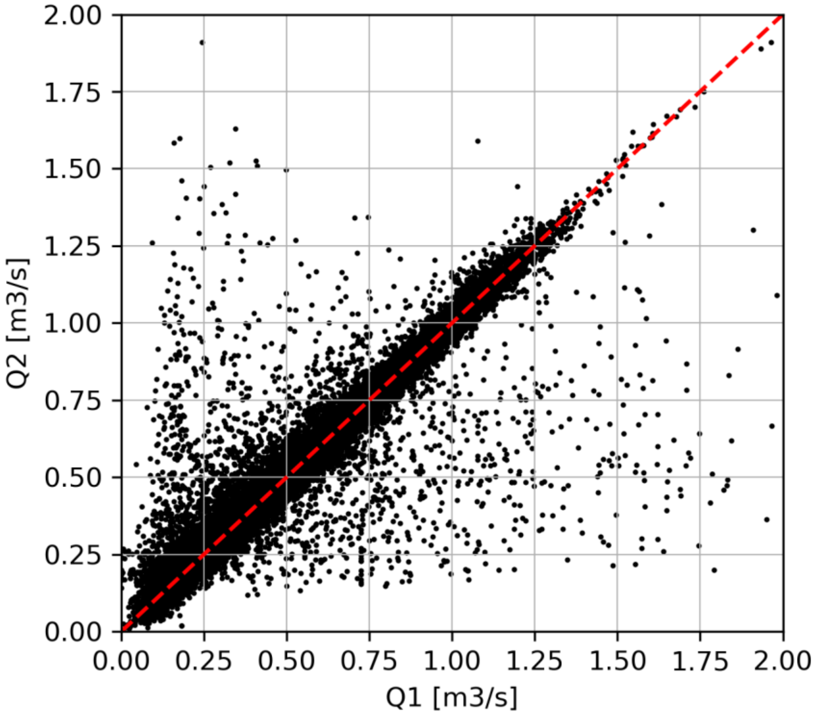


Figure 31: Comparison of concurrent flowrates discharged from Outlet A (Q1) and B (Q2).

The combined discharge flow rate for the years 2023 and 2024, calculated as the sum of the flowrates discharged through the two pipelines, is presented in Figure 32. For future reference, two (2) four-weeks' long periods, centred around dates when a member of the community raised his concerns about algae on the beach and coastal areas, are highlighted in red. The total flow rates discharged over these two periods are show in greater detail in Figure 33, which reveals a distinct daily pattern. Figure 34 depicts the hourly median discharge flowrate during the summer months. Maximum discharge volumes of approximately 1.05 m^3/s typically occur at 11 pm (local time), with a secondary peak of about 0.70 m^3/s observed early in the afternoon at 2 pm. The lowest discharge flow rates are generally recorded between 4 and 10 am. Discharge patterns in winter are similar but exhibit higher flowrates.

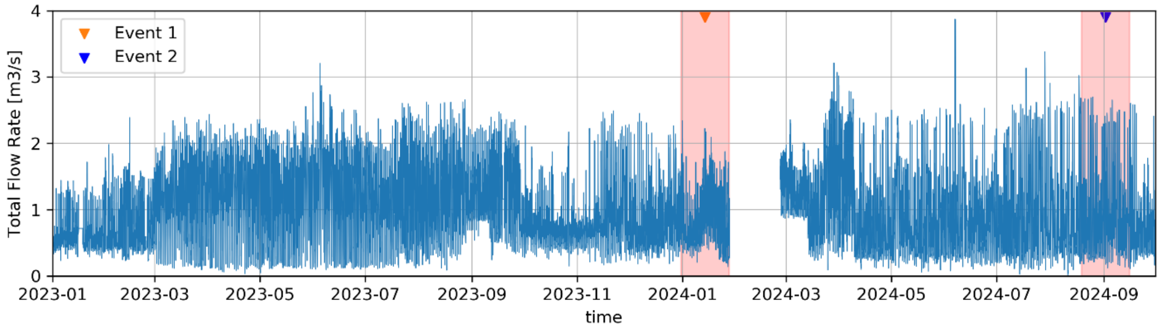


Figure 32: Total flow rate time series for 2023 – 2024 [source: WaterCorp]. The timings of two algal bloom events reported by the public are demarked with triangles. The shaded portions of the figure are expanded in Figure 33.

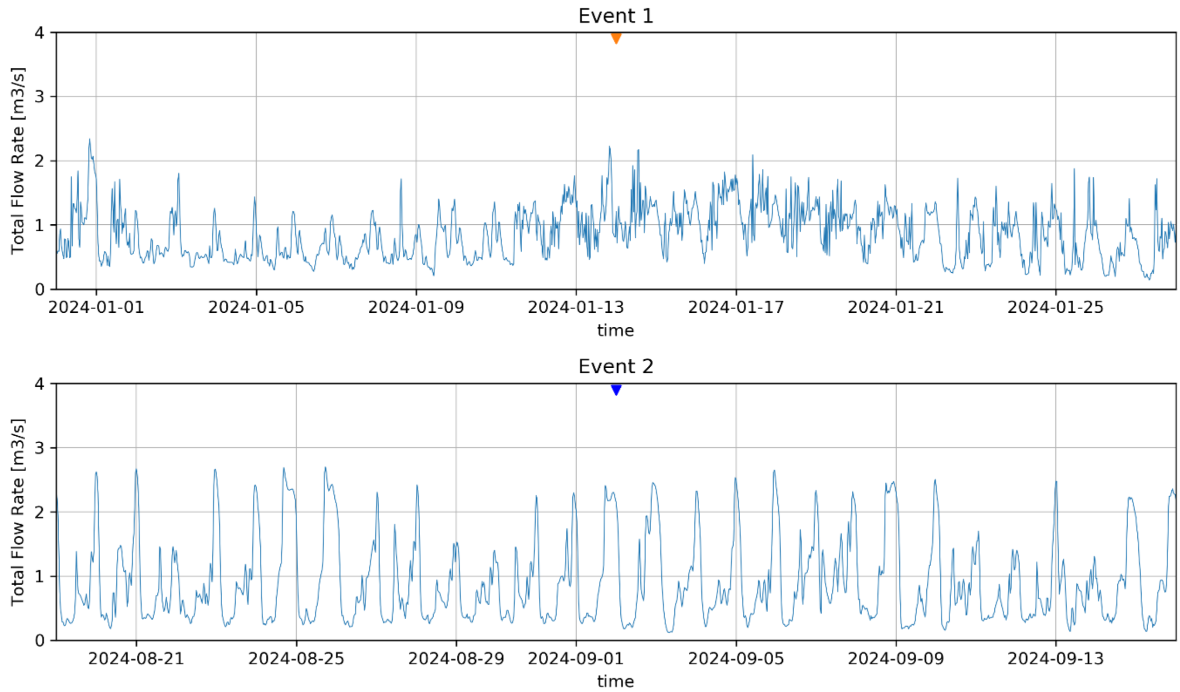


Figure 33: Four-week's of combined discharge rates zoomed in around two days of concern to the community in January 2024 (top) and August/September, 2024 (bottom).

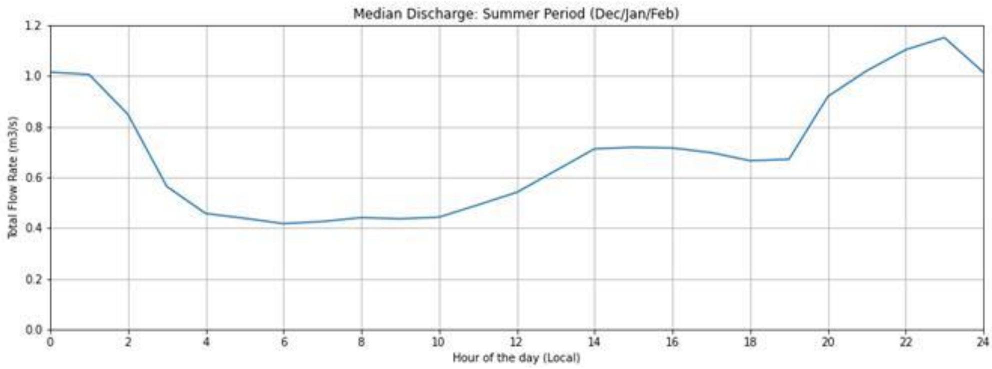


Figure 34: Discharge daily pattern derived from the summer months data. Winter conditions can be described with a similar pattern, although it exhibits higher flow rates.

The probability density distributions of the discharge rates are shown in Figure 35.

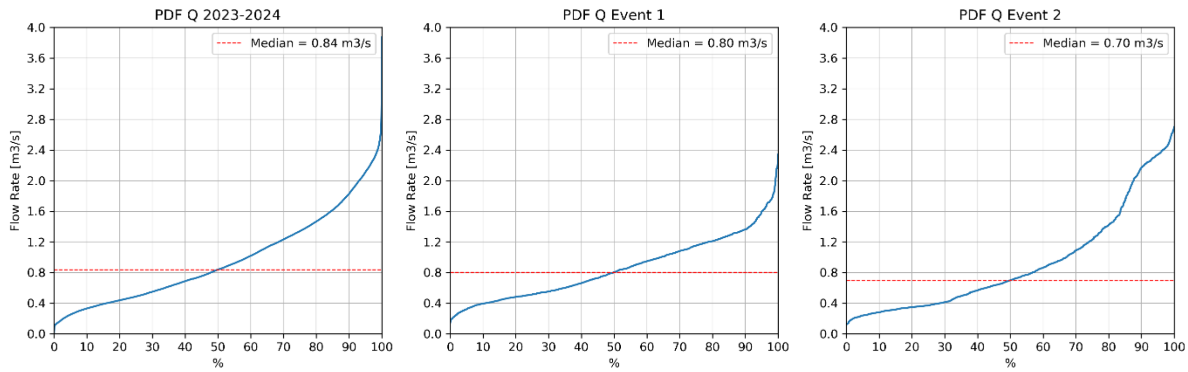


Figure 35: Probability density distributions of the combined effluent flow rate over the full period provided (left), during the four weeks around event 1 (centre), and event 2 (right). Median flow rates are demarked with red dash lines.

Pattiaratchi (1991) described the density of the Beenyup effluent as 1,000 kg/m^3, placing it within the low salinity 'freshwater' category. Weekly readings for conductivity support this characterisation (Section 3.3.2.2). Assuming the freshwater discharge presents a salinity content of 0 PSU, the UNESCO (1981) equation of state can be used to determine the water density from temperature, salinity, and pressure, suggesting that discharge temperatures between 4° to 14°C would result in minimal changes to density and therefore would maintain the expected behaviour of the effluent plume in seawater: the positive buoyant plume tends to raise to the surface. Here, a discharge temperature of 10°C was adopted.

4.3. Near-Field Mixing: Theoretical Model

Key parameters adopted in the theoretical near-field mixing model consisted of:

- Water depth = 10 m
- Background density = 1025.0 kg/m^3, derived from the UNESCO 1981 equation of state for a salinity of 36.2 PSU, temperature 22.5°C, and hydrostatic pressure of 1 bar, corresponding to summer conditions.
- Flow rate = 0.84 m^3/s (the median flowrate of the combined discharge, refer Figure 35)
- Number of ports = 98
- Port diameter = 0.16 m
- Effluent density = 999.7 kg/m^3 (0 PSU, 10°C, 1 bar)

Positively buoyant discharges carrying initial momentum are known as buoyant jets. From the point of discharge and over a distance, z, mixing and dispersion respond to jet-like principles. Far from the discharge point, plume-like characteristics dominate. According to Fisher et al (1979), a buoyant jet will behave as a jet when $z \ll l_m$, where l_m is the momentum length scale, and exhibit a plume-like behaviour where $z \gg l_m$. The length scale l_m can be calculated as follows:

$$l_m = \frac{M^{3/4}}{B^{1/2}}$$

Where $M_o = Qw$ is the initial momentum flux (m⁴/s²) and $B = g(\Delta\rho/\rho)Q$ is the initial buoyancy flux (m⁴/s³), both at the point of discharge. In these equations, Q is the volume flux (also referred to as flow rate, in m³/s), w the mean discharge velocity of the jet (m/s), g the acceleration due to gravity (m/s²), ρ the density of the seawater (kg/m³), and $\Delta\rho$ the density gradient between the effluent and seawater (kg/m³). Solving for the parameters listed above:

- $Q = \frac{0.84}{98} = 0.00857$ m³/s (flow rate per port)
- $w = \frac{4Q}{\pi d^2} = 0.43$ m/s (mean velocity through each port)
- $M_o = 3.65 \times 10^{-2}$ m⁴/s²
- $B = 9.81 \cdot (1025 - 999.7) \cdot \frac{0.00857}{1025} = 2.07 \times 10^{-2}$ m⁴/s²

The characteristic length scale, $l_m = 0.33$ m, is much shorter than the distance available for mixing ($z = 10$ m) and, therefore, it can be assumed that the buoyant jet behaves as a simple plume through most of its ascent through the water column.

The volume flux (μ) of a simple plume increases as surrounding seawater is entrained during its ascent to the surface. Assuming the ascent occurs in stagnant, homogenous waters, the volume flux may be estimated with Fisher et al.'s (1979) equation 9.30, such that $\mu = 0.15\, B^{1/3}\, z^{5/3} = 0.89$ m³/s (for z = 10 m). This theoretical model suggests that the plume will be diluted by a factor of 10 ([6]) by the time it reaches the water surface.

4.4. Grid Cell Size Requirements for the Far-Field Model

It can be argued that near-field dilution processes do not terminate at the point of impingement, as the plume must spread radially at the surface with a velocity v, to accommodate for the diluted effluent that continuously reaches the water surface. However, mixing of the surface layer is primarily governed by surface layer dynamics and wind effects. Recognising that near-field mixing processes occurring at sub-grid cell scales can only be represented through numerical diffusion, the maximum horizontal grid cell size at the diffuser site was determined based on the chosen vertical grid cell size, the previously derived near-field dilution, and the tentative far-field modelling timestep.

Conductivity, temperature, depth (CTD) profiles gathered near the Ocean Reef Marina indicate that the water column is not perfectly homogeneous, and that mild thermal stratification exists (Section 2.5). O2Me analysis of these CTD profiles indicated that a far-field model discretised with three (3) vertical layers of equal thickness should provide sufficient vertical resolution to capture the observed stratification. At a water depth of 10 m representative of the conditions at the Beenyup effluent site, each layer would be approximately 3.3 m high.

Preliminary simulations indicated that a modelling timestep of 30 minutes would provide the necessary temporal resolution while ensuring computational efficiency and maintaining a manageable modelling runtime. By multiplying the modelling timestep and volume flux, μ, of the diluted plume at the surface, a volume of diluted plume equal to 1,600 m³ ([7]) was obtained. Neglecting other near-field mixing

[6] (0.89 / 0.00857) = 10.4
[7] (0.89 * 30 * 60) = 1,602

processes and considering the 3.3 m thick layer at the surface, the theoretical 10-fold dilution requires a cell size 10-times larger than the volume calculated. It follows that the maximum cell size shall have a horizontal length scale $Hcell < 70$ m.

The minimum grid cell scale was extrapolated from Fisher et al (1979), table 9.1, which presents experimental results related to the width of a simple plume as a function of the distance from the discharge point, *z*. When $z \gg l_m$, these experimental results suggest that the width of the plume is approximately equal to 10% of *z*, yielding a radius *r* ~1 m at the water surface. In simple terms, $Hcell > 2$ m.

For practical reasons related to accurately positioning the diffuser nozzles, and given that smaller cell sizes increase the numerical accuracy of the mixing and dispersion processes modelled, the horizontal length scale of the cells in the vicinity of the diffuser was set to $Hcell = 12$ m.

5. Far-field Model Development

The far-field model was developed through an iterative process, evaluating model performance across a range of tidal and wind forcings, as well as various numerical mesh configurations and vertical discretization schemes (2D and 3D). Priority was given to the model's ability to reproduce wind driven drift, while tidal currents were also considered.

5.1. Numerical Software

Far-field modelling was conducted using DHI's MIKE FM three-dimensional (3D) suite of models (DHI 2024). The MIKE FM hydrodynamic (HD) module solves the unsteady Reynolds-averaged Navier Stokes (RANS) equations (mass and momentum) and can be fully coupled with a surface wave-action equation. Spatial discretisation uses a finite volume approximation on unstructured horizontal grids with three-sided polygons. The vertical coordinate in the hydrodynamic module can be discretised with either a sigma or combined sigma-z scheme. Alternately, the depth averaged equations can be solved without vertical discretisation.

The model incorporates non-hydrostatic and baroclinic pressure, though hydrostatic and barotropic assumptions can be used for increased efficiency and stability. Coriolis may be specified as fully variable, or by f-plane approximation. A two-equation ($\boldsymbol{k}-\boldsymbol{\epsilon}$) closure scheme is implemented for both horizontal and vertical eddy viscosities, or alternately a Smagorinsky formulation may be used in the horizontal. If required, air-sea fluxes, tidal potential and ice-coverage may be prescribed.

The MIKE suite of models incorporates several additional modules for transport of dissolved or suspended material. Coupled sub-grid near-field models are included (e.g. for ocean outfalls and sea-dumping), and far field diffusivities scale on the eddy viscosity from the hydrodynamic module.

The advection/dispersion module adopted in this study simulates the spreading of dissolved substances subject to advection and dispersion processes. The module can be applied to a wide range of hydraulic phenomena and water quality studies, including spreading of dissolved substances such as salt, heat, bacteria and xenobiotic compounds. The module solves the mass-conservation equation for dissolved or suspended substances. Discharge quantities and compound concentrations at source and sink points are included together with a decay rate. The information of water velocities and depth needed in the equation are obtained from hydrodynamic module.

5.2. Model Bathymetry

The model bathymetry was compiled from multiple data products. Datasets were prioritised as follows:

1. High-resolution Lidar and Multibeam observations along Perth coasts with spatial resolution ranging from 1m-10m provided by DOT. The highest resolution of 1 m is located at Ocean Reef coastal area.

Geoscience Australia (GA) 50m Multibeam Dataset of Australia 2018.

Geosciences Australia 250 m gridded bathymetry product.

Bathymetric datasets were merged in the MIKE Zero bathymetry interpolation tool according to the established prioritization.

The areal coverage of each dataset relative to the different modelling domains considered in Section 5.3, is presented in Figure 36.

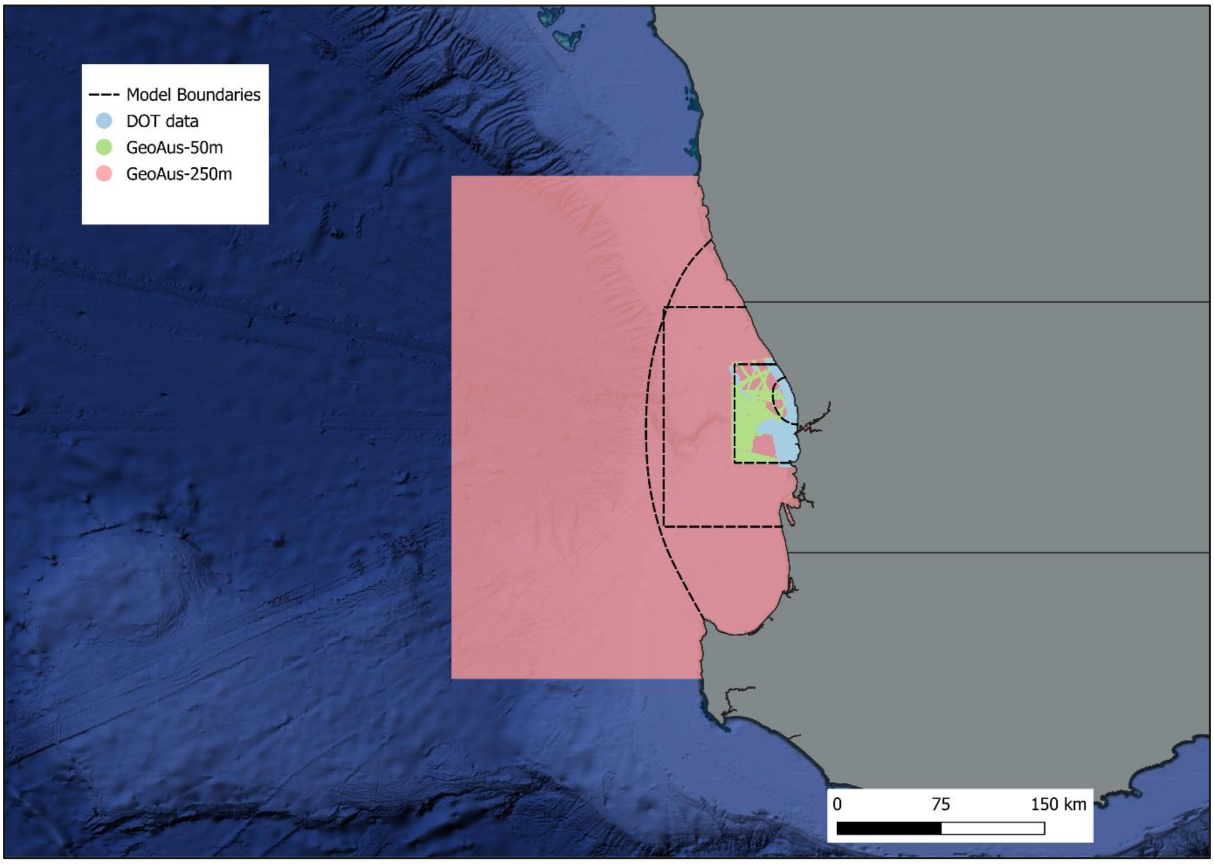


Figure 36: Extents of bathymetric datasets considered for the far-field model

Following the compilation of the bathymetric datasets, spot validations of water depths were performed by comparing water depths as recorded by AWAC sensors and surface elevation recordings at Fremantle Port against compiled bathymetric values. Deviations were found to be less than 2%, confirming the suitability of the compiled bathymetry for this study.

5.3. Numerical Domain

The analysis of the optimal numerical mesh (or computational grid) configuration centred on identifying the set up that would best replicate the current field observed at the study site. Four model domains were considered for this study, each varying in size and numerical resolution. Tests considered single isolated grids as well as nested configurations. Optimal results were obtained with Mesh A nested into Mesh D and were intimately linked to the selection of the wind forcing and wind friction coefficients, as it will be described in later sections of this report. Details of the two numerical meshes are provided in Section 5.5.

The areal extent of each domain considered is shown in Figure 37. Numerical characteristics are listed in Table 18.

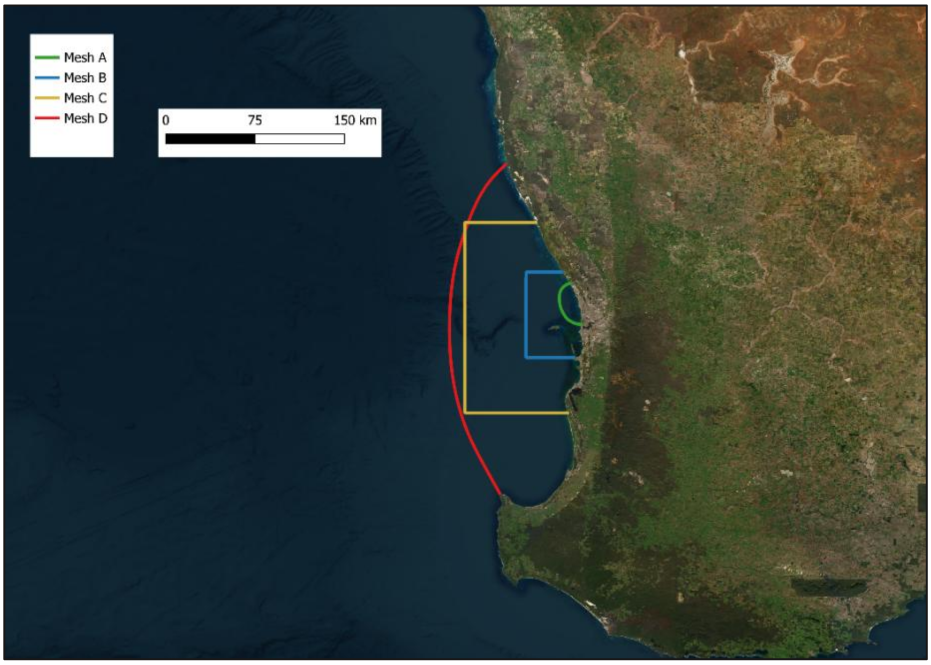


Figure 37: Location of outer boundary for meshes A, B, C, and D.

Table 18: Numerical grids considered for the study

Mesh	Offshore Boundary		Element length scale		Intent
	Distance from Project	Depth	Max	Min	
A	16 km	40 m AHD	800 m	12 m	High resolution model nested into larger domains, forced with modelled water levels at its open boundary and local winds at its surface.
B	45 km	140 m AHD	1700 m	15 m	High resolution in the local and regional area, forced with water levels extracted from global models and winds from either global models or local measurements.
C	100 km	1100 m AHD	2200 m	20 m	High resolution in the local and regional area, forced with water levels extracted from global models and winds from either global models or local measurements.
D	200 km	3500 m AHD	9000 m	20 m	A coarse-resolution model designed to replicate large-scale hydrodynamics, primarily intended for deriving water levels at the boundaries of nested models.

5.4. Key Inputs to the Far-Field Model and Sensitivity Tests

The models were forced with tidal water levels at the open boundaries and wind at the surface, which are the major driving forces for currents (and therefore mixing and dispersion) in the area (Pattiaratchi, 1991). Waves, storm surges, seiches, surface heat-exchange with the atmosphere, groundwater inflows and rain were therefore neglected.

5.4.1. Water Levels

The open boundaries of Meshes B, C and D were driven by water-levels reconstructed from the harmonic constituents extracted from the TPXO Indian Ocean model, available with 1/12th-degree resolution. Spatial interpolation of the TPXO outputs onto the boundary points was performed by a weighted nearest neighbour interpolation. For each boundary node the three nearest TPXO grid points were used, weighted by the inverse of the distance between TPXO output location and the boundary node location.

The open boundary of Mesh A was forced with water levels extracted from the simulations conducted using Mesh D (nesting).

5.4.2. Wind Forcing

The dominant influence of local winds on the current field at Ocean Reef required a thorough assessment of available wind sources and an evaluation of which one to select for the study. Two data sources were considered:

- European Centre for Medium Range Weather Forecasts (ECMWF): Wind and atmospheric pressure from the ERA5 0.25 degree hourly hindcast dataset (global hindcast model).
- Bureau of Meteorology (BOM): Stations at Ocean Reef, Rottnest Island and Swanbourne (refer Section 2.2).

At BOM's station sites, BOM's data were preferred over ECMWF's data. To assess ECMWF's accuracy at these sites and determine whether the spatially varying wind field applied to Meshes B, C, and D was accurate, the two wind datasets were compared. First, wind speed and direction from the ERA5 model were interpolated at the BOM station sites. Second, the wind data for the year 2019 were compared, as 2019 was readily available for the comparison and aligned with the timing of the construction activities and observational datasets.

Figure 38 to Figure 44 compare the ERA5 winds interpolated at the BOM station sites with BOM winds during the summer months, a critical period for this study due to the increased likelihood of algal blooms in warm weather and waters. Close alignment of ERA5 to BOM wind direction measurements is observed in all cases, however some differences in wind intensity are evident. To quantify the error in wind intensity, wind speed correlation plots were prepared (Figure 41) which revelated that ERA5:

- Underestimates wind speeds at Ocean Reef by ~ 25%.
- Overestimates wind speeds at Swanbourne by ~ 10-15 %.
- Is sufficiently accurate at Rottnest Island.

The same comparison was also conducted for winter conditions (Figure 42 to Figure 45), yielding equivalent results.

From the ERA5 to BOM wind station comparison, it was concluded that ERA5 provides an appropriate spatial representation of the wind direction around the study site, and suitable wind intensity offshore. However, local wind intensities may not be best replicated with ERA5 winds.

The sensitivity of local hydrodynamic circulation to different wind sources and domains was tested for the following cases:

1. Mesh B forced with ERA5 wind field.
2. Mesh C forced with ERA5 wind field.
3. Mesh D forced with ERA5 wind field.
4. Mesh D forced with ERA5 wind field, uniformly scaled over the full domain by +25%.
5. Mesh D forced with ERA5 wind field, scaled by +25% at the Ocean Reef node, and -10% at the Cottesloe (closest to Swanbourne) node, all other nodes not scaled.
6. Mesh A forced with BOM's Ocean Reef winds applied uniformly over the full mesh.

In cases 1 to 3, the intent was to assess the influence of domain size for a spatially variable wind. The right hydrodynamic trends were captured, as demonstrated by the comparison of current intensity outputs at the Ocean Reef AWAC sites, but directionality was sub-standard. Overall, Mesh D performed better than the others.

Cases 4 and 5 tested the influence of the wind intensity on a large domain. Results from these tests were compared to those from Case 3. The abrupt reduction in the overall wind fields returned sub-standard results.

Case 6 focused on numerical resolution (cell size) and was designed to replicate the conditions observed at Ocean Reef. Given the relatively small domain and its intended purpose, this configuration was forced with spatially uniform winds from the BOM Ocean Reef station without any scaling. This setup produced the best results compared to measured currents. However, hydrodynamic reliability near Mesh A boundaries is expected to decrease due to:

- The absence of momentum fluxes at the open boundary of Mesh A, where only water levels from Mesh D were applied.
- The use of a uniform wind field.

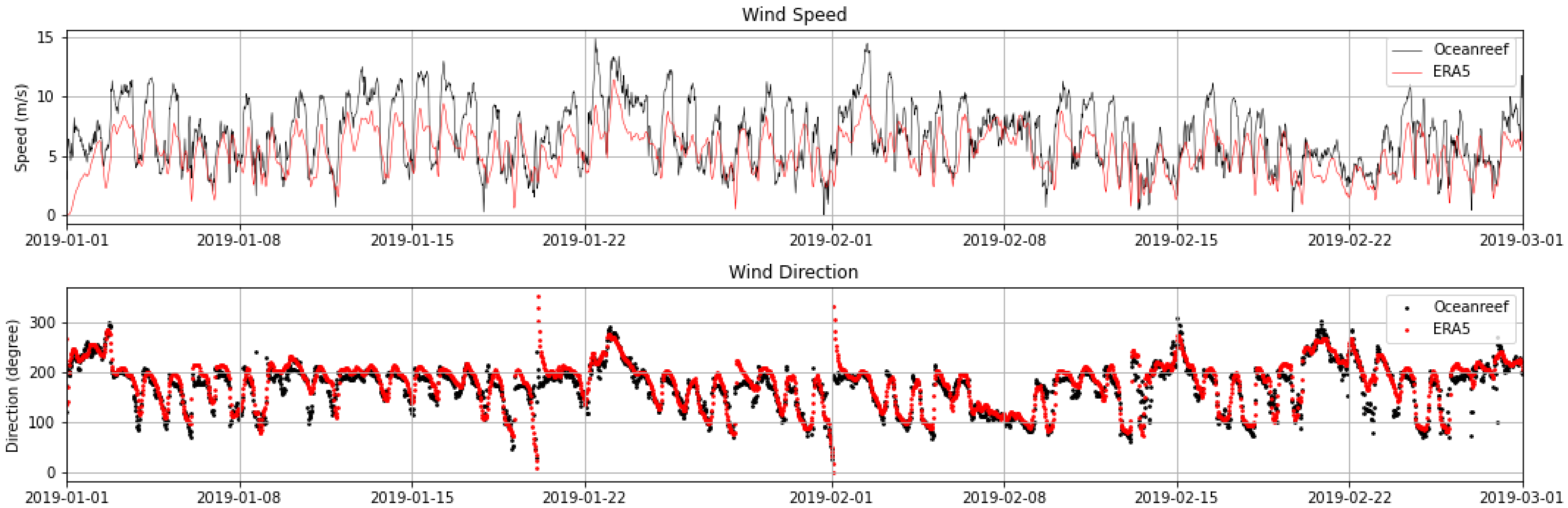


Figure 38: Wind Speed and Direction at Ocean Reef during Summer (January & February) 2019: ERA5 extraction and BoM observations

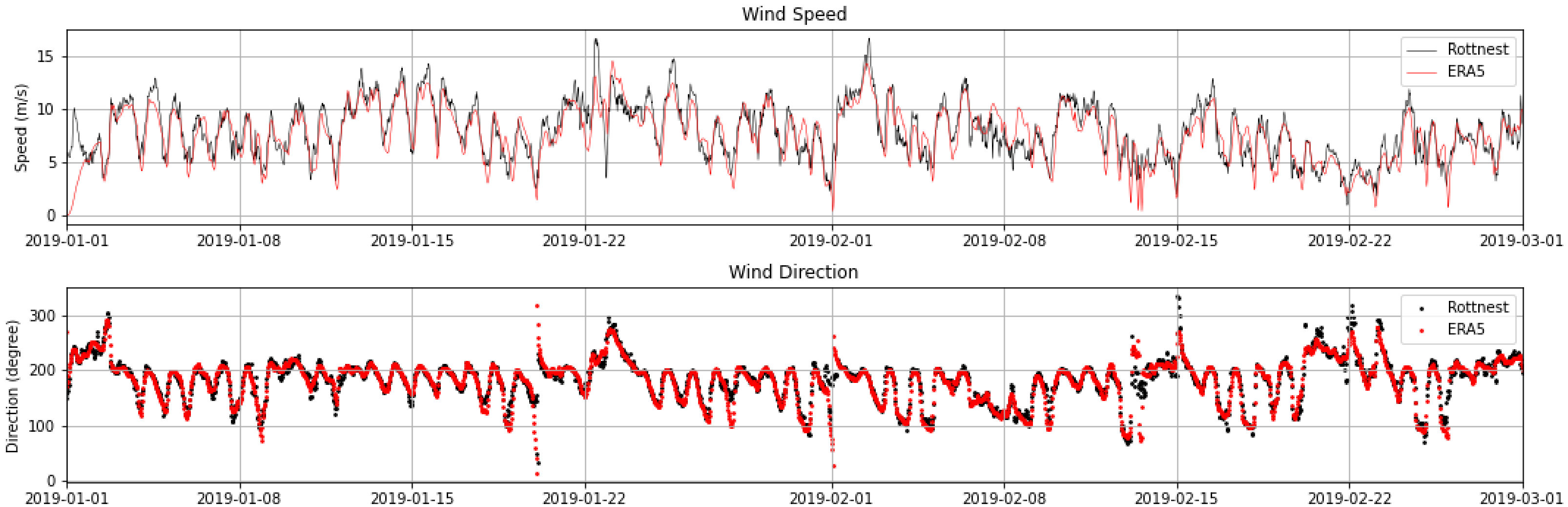


Figure 39: Wind Speed and Direction at Rottnest Island during Summer (January & February) 2019: ERA5 extraction and BoM observations

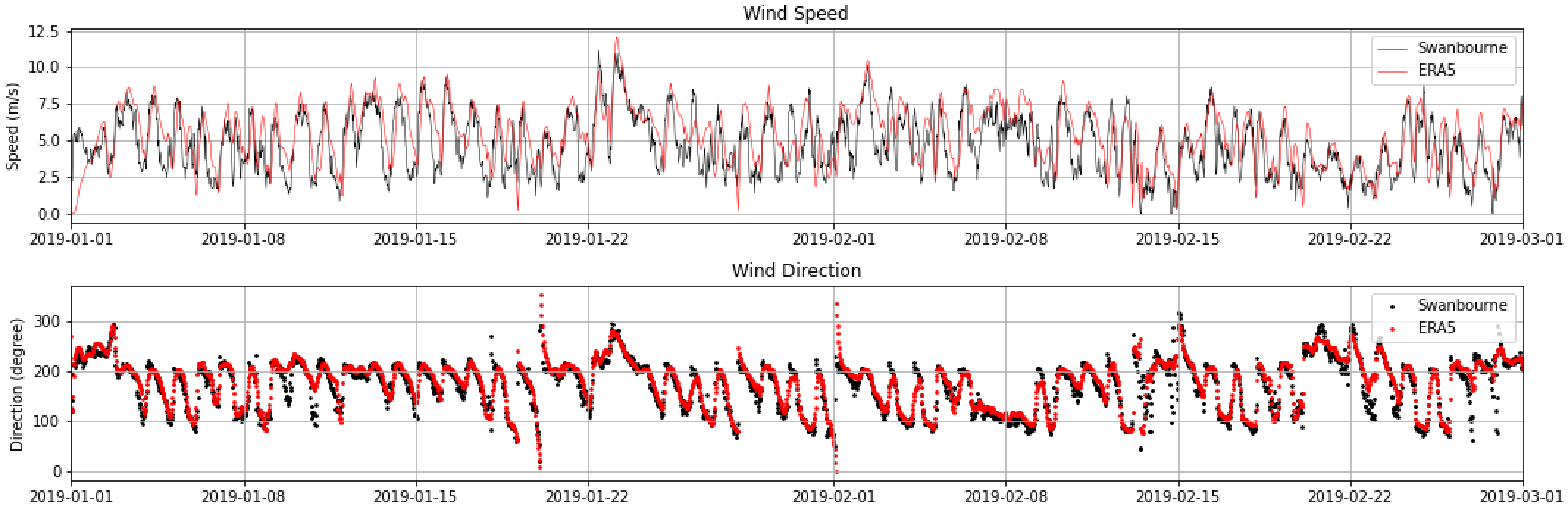


Figure 40: Wind Speed and Direction at Swanbourne during Summer (January & February) 2019: ERA5 extraction and BoM observations

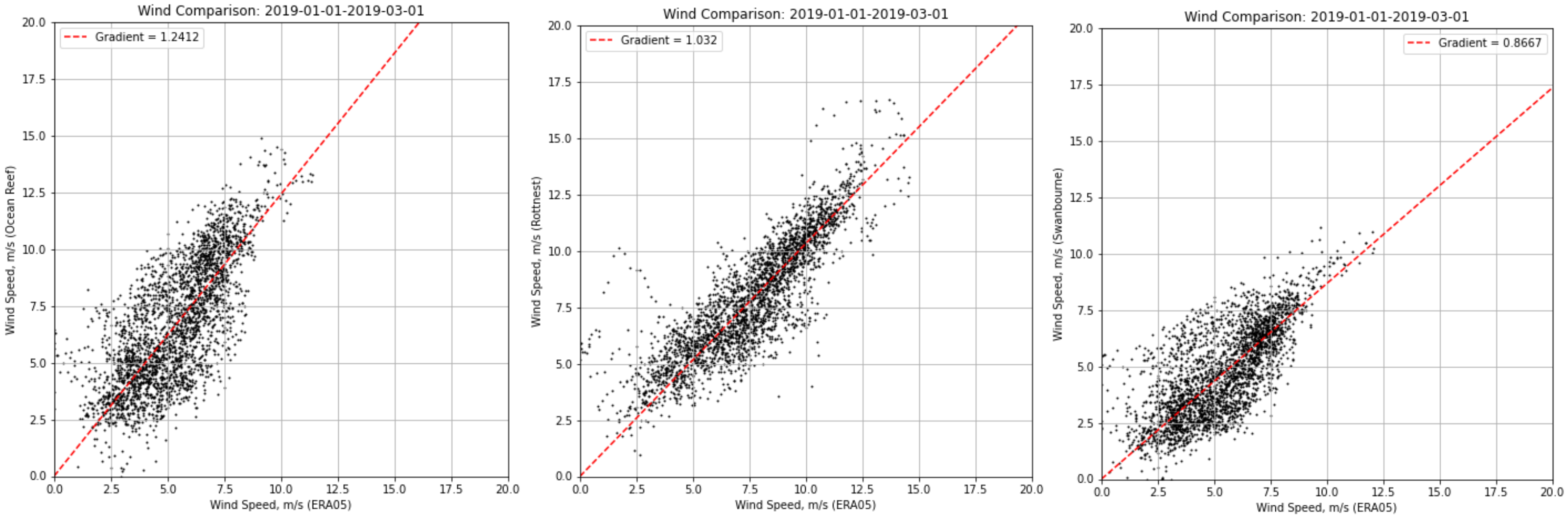


Figure 41:ERA05 correlation with BoM Stations (left = Ocean Reef, middle = Rottnest, right = Swanbourne) for Summer (January & February) 2019.

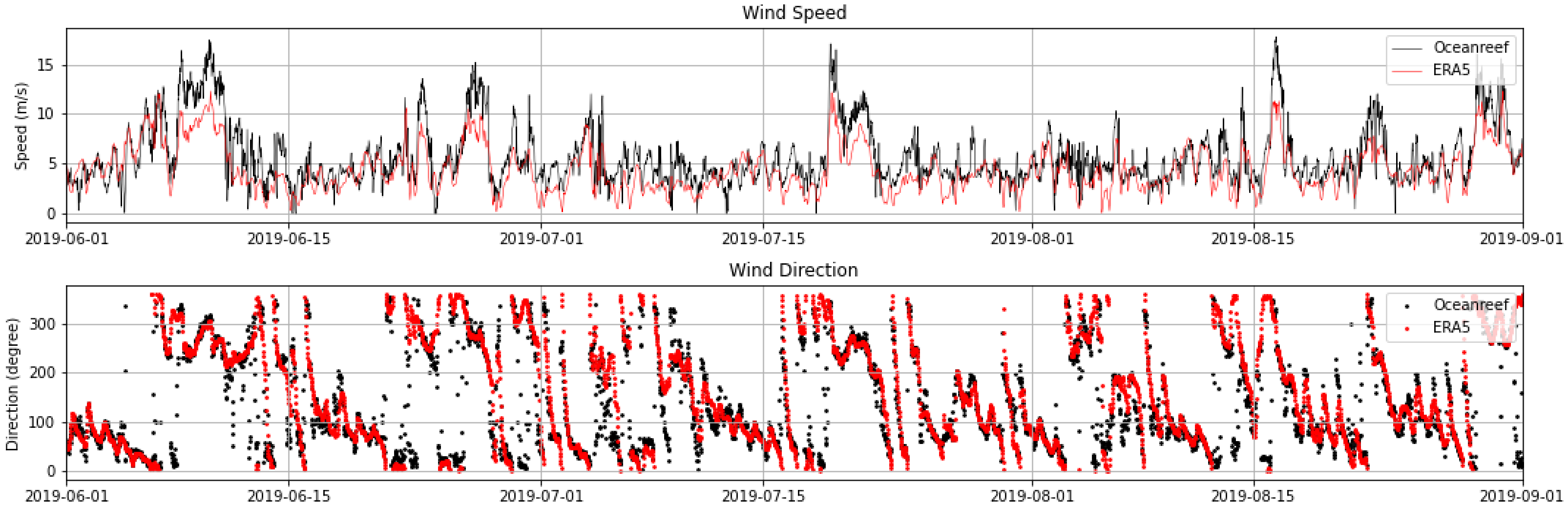


Figure 42: Wind Speed and Direction at Ocean Reef during Winter (June, July & August) 2019: ERA5 extraction and BoM observations

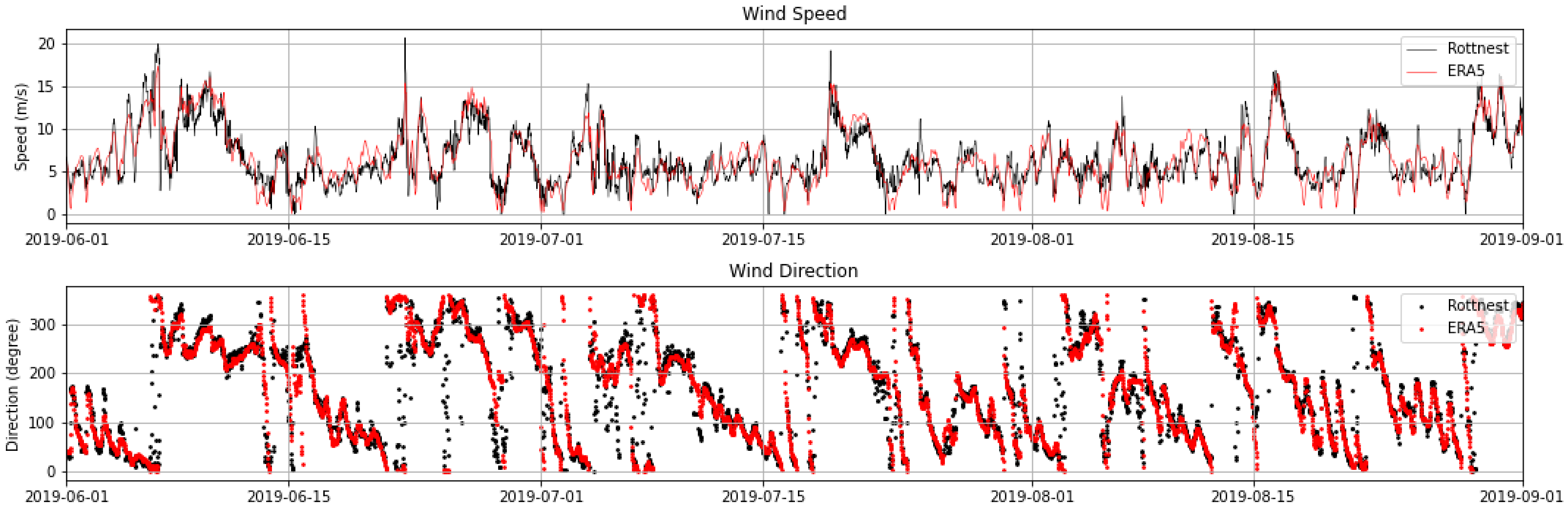


Figure 43: Wind Speed and Direction at Rottnest Island during Winter (June, July & August) 2019: ERA5 extraction and BoM observations

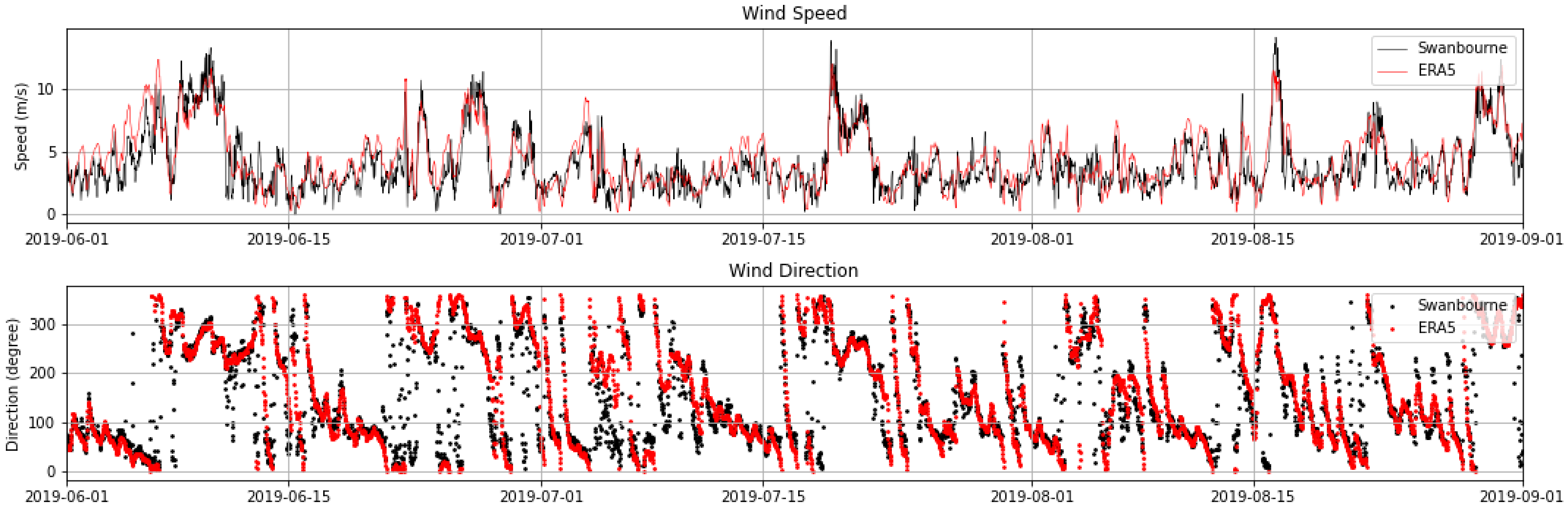


Figure 44: Wind Speed and Direction at Swanbourne during Winter (June, July & August) 2019: ERA5 extraction and BoM observations

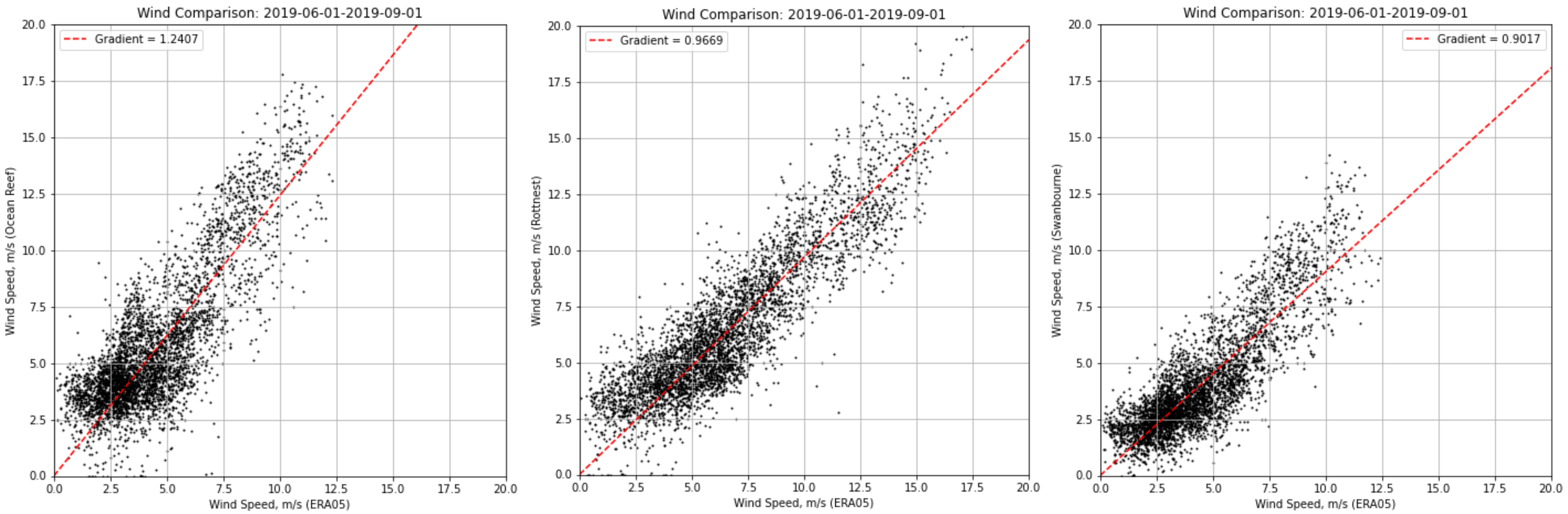


Figure 45: ERA05 correlation with BoM Stations (left = Ocean Reef, middle = Rottnest, right = Swanbourne) for Winter (June, July & August) 2019.

5.4.3. Wind Friction Model

As the localised hydrodynamics surrounding Ocean Reef are largely controlled by local winds (Section 2.4), the selection of an appropriate surface stress model is of great importance. Wind stress ($\bar{\tau}$) at the water surface is parameterised with the following equation:

$$\bar{\tau} = \rho_a C_d |U_w| \overline{U_w}$$

Where ρ_a is the density of air, U_w is the wind speed 10 m above the sea surface, and C_d is the wind drag coefficient which can be specified as constant or linearly varying as a function of wind intensity.

The sensitivity of local hydrodynamic circulation to a range of wind drag coefficients was tested for 2D (depth averaged) conditions and two seasons: summer and winter. Simulations were run for Mesh A.

The following typical drag coefficients were tested:

1. C_d = 0.00125 (constant)
2. C_d = 0.00175 (constant)
3. C_d = 0.00200 (constant)
4. C_d = 0.00250 (constant)
5. C_d = 0.00125 at wind speed of 5 m/s, linearly varying to C_d = 0.00250 at 20 m/s
6. C_d = 0.0014 at wind speed of 0 m/s, linearly varying to C_d = 0.0026 at 24 m/s
7. C_d = 0.00063 at wind speed of 0 m/s, linearly varying to C_d = 0.00723 at 100 m/s

Tests 1 to 4 yielded unsatisfactory results, redirecting the attention to variable wind friction coefficients. Test 6 provided the best comparison to measured currents in summer but overestimated wind-driven currents during winter. Results from Test 5 compared well for winter conditions but underestimated the summer currents. Test 7 returned a balanced outcome between summer and winter hydrodynamics, probably owing to its wider range of coefficients which suited the local conditions.

5.4.4. Horizontal Eddy Diffusivity

Horizontal dispersion plays a key role in the spreading of substances in the ocean, which is the focus of this study. The formulation of the horizontal dispersion model involves the selection of a turbulent diffusion model and its empirical parameters. Typically, these parameters are either derived from field measurements (e.g., by sampling the dilution of rhodamine dye release or the spreading of drift drogues) or adopted as a calibration tool by comparing modelling results to spatial and temporal measurements substances found in the water body (e.g., nutrients, heavy metals, etc.). Due to time constraints, neither approach was feasible; therefore, the selection of a suitable dispersion model (and its key parameter) was based on previous studies undertaken in the area.

Steedman and Associates (1976) combined data gathered during the early design process for the Beenyup outfall in 1976 with length scale arguments (refer Okubo, 1974; Fisher et al 1979) to propose a horizontal eddy diffusion coefficient in the range 0.5 – 5.0 m^2/s. Since horizontal eddy diffusivity scales with grid size, Hillmer and Imberger (2007) used a value of 5.0 m^2/s for numerical cell sizes of 250 m in an ecological modelling of the Marmion Marine Park. Their choice was also informed by earlier studies by Pattiaratchi and Knock (1995) and Zaker (1998). A similar scaling argument was adopted by Apasa

(2011), who, through a model calibration exercise comparing model to dye concentrations, arrived at a horizontal eddy diffusion coefficient of 0.1 m^2/s for their finest model with ~12 m grid cells.

O2Me considered both horizontal eddy diffusivity and dispersion coefficient applications for this study. The former as an estimate of turbulent mixing due to sub-grid scale eddies in the horizontal plane, while the latter to represent the combined effects of advection and mixing over larger scales (order of 10^2 to 10^3 m). In this study, a constant horizontal eddy diffusion coefficient of 0.1 m^2/s was adopted.

5.5. Model Configuration and Parameters

The extensive testing conducted helped identify the optimum model configuration for validation:

- Mesh A (Figure 46) was selected to investigate the effects of the Ocean Reef Marina on localised currents and the potential changes in the distribution of the Beenyup effluent compared to pre-redevelopment conditions.
- Mesh D was forced with a tidal signal reconstructed from harmonic constituents extracted from the TPXO Indian Ocean model.
- Mesh D was run in 2D while Mesh A (Figure 47) was executed in 3D using a vertical sigma-layer discretisation with three layers of equal thickness.
- Mesh A was nested into Mesh D. Water levels extracted from Mesh D were applied at the open boundary of Mesh A.
- Mesh D was forced with a spatially variable wind field (ERA5), without any scaling.
- Mesh A was forced with a uniform wind field as measured at BoM's Ocean Reef station.
- Wind energy was transferred to the surface layer of both Mesh A and D using a wind drag coefficient that varied linearly with wind speed, from 0.00063 at 0 m/s to 0.00723 at 100 m/s.
- A horizontal eddy diffusion coefficient of 0.1 m^2/s and vertical 10^{-5} m^2/s were adopted.
- Two versions of Mesh A and Mesh D were created, one to represent pre- and the other post-development conditions (Figure 46).
- A quadrangular grid with 15 m x 15 m cell resolution around the Beenyup effluent and extending to the LEPA boundaries was incorporated into all four numerical grids (Mesh A and B, for pre- and post-configurations) (Figure 47).
- Ambient and effluent temperatures and salinities for summer conditions applied to all modelled scenarios are shown in Table 19.
- The effluent was represented through the introduction of a dimensionless numerical signal discharged with an initial concentration of 1 into an environment with a concentration of 0.
- The Beenyup effluent was discharged at the bottom layer of Mesh A, through ports evenly distributed along the diffusers, with a variable flow rate that aligned with the typical summer discharge pattern (Figure 34).

Table 19: Modelled ambient and discharge temperature and density (Section 4.3)

Parameter	Modelled Ambient	Modelled Discharge
Temperature	22.5 °C	10°C
Salinity	36.2 PSU	0 PSU
Numerical Signal Concentration	0	1

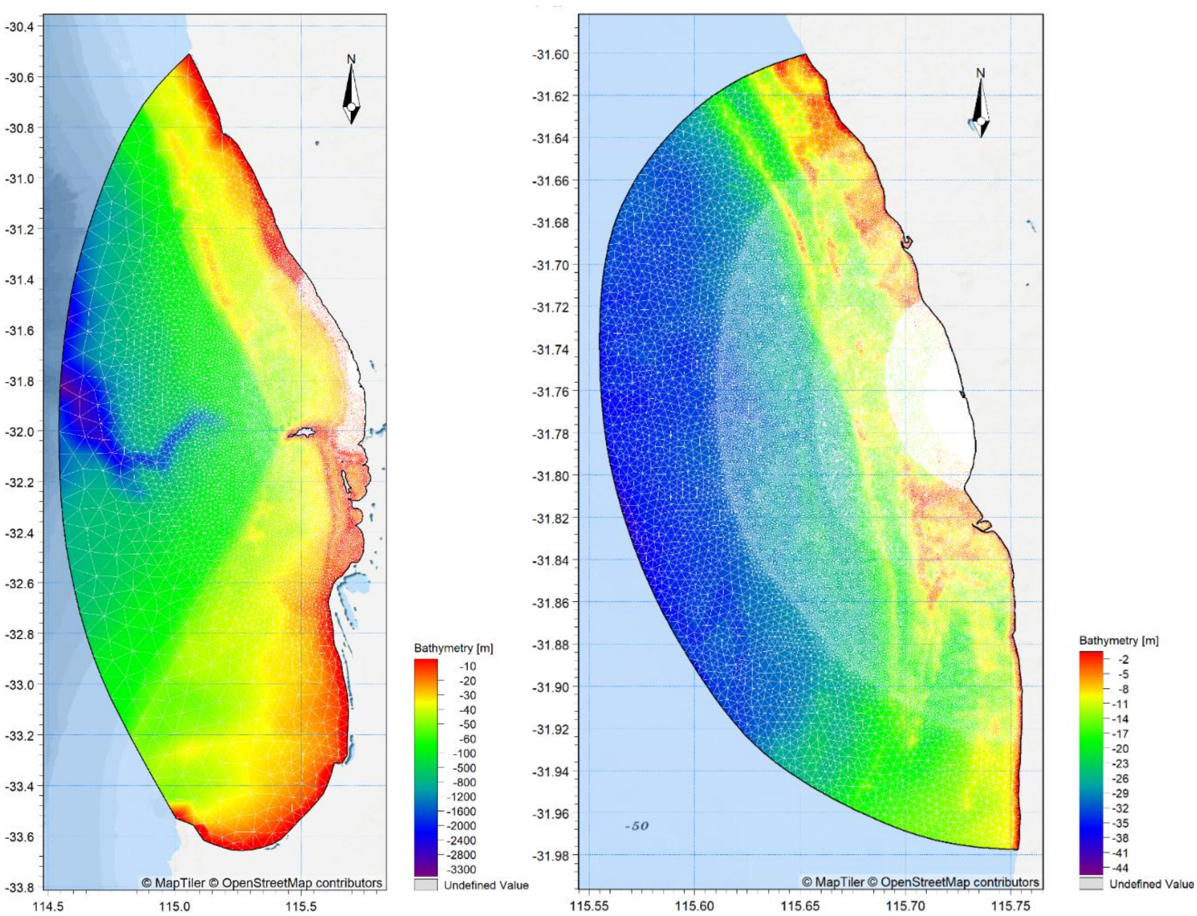
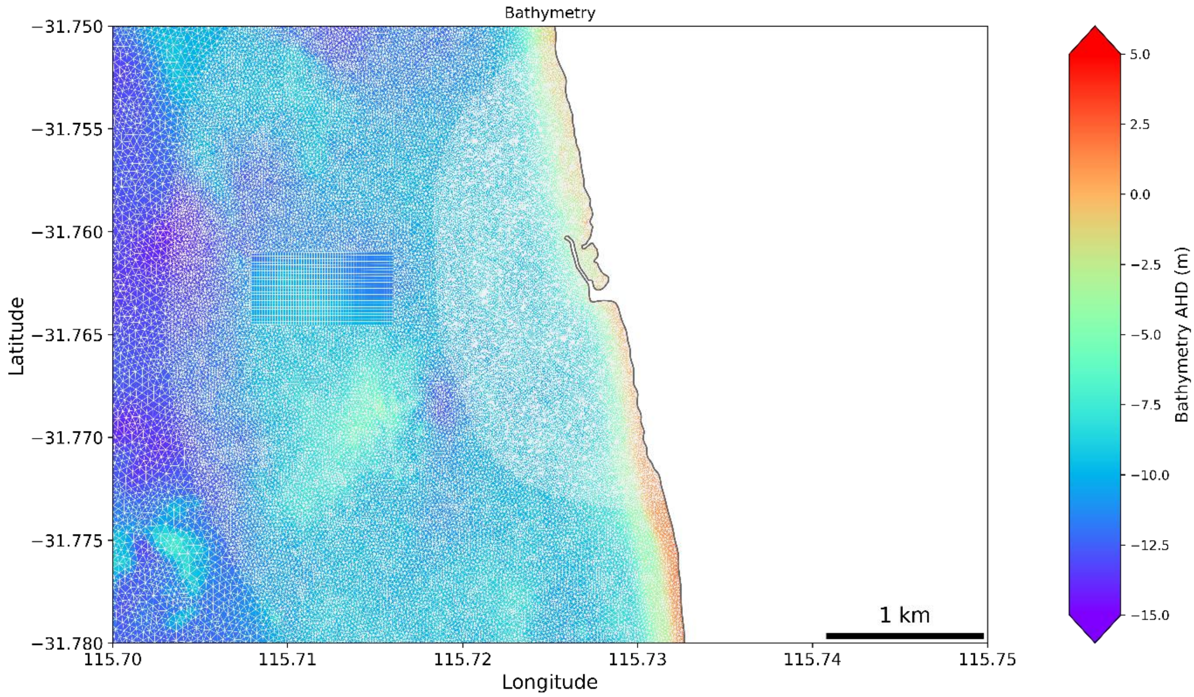


Figure 46: Numerical Meshes and bathymetry. Left = Regional Mesh D, right = Local Mesh A



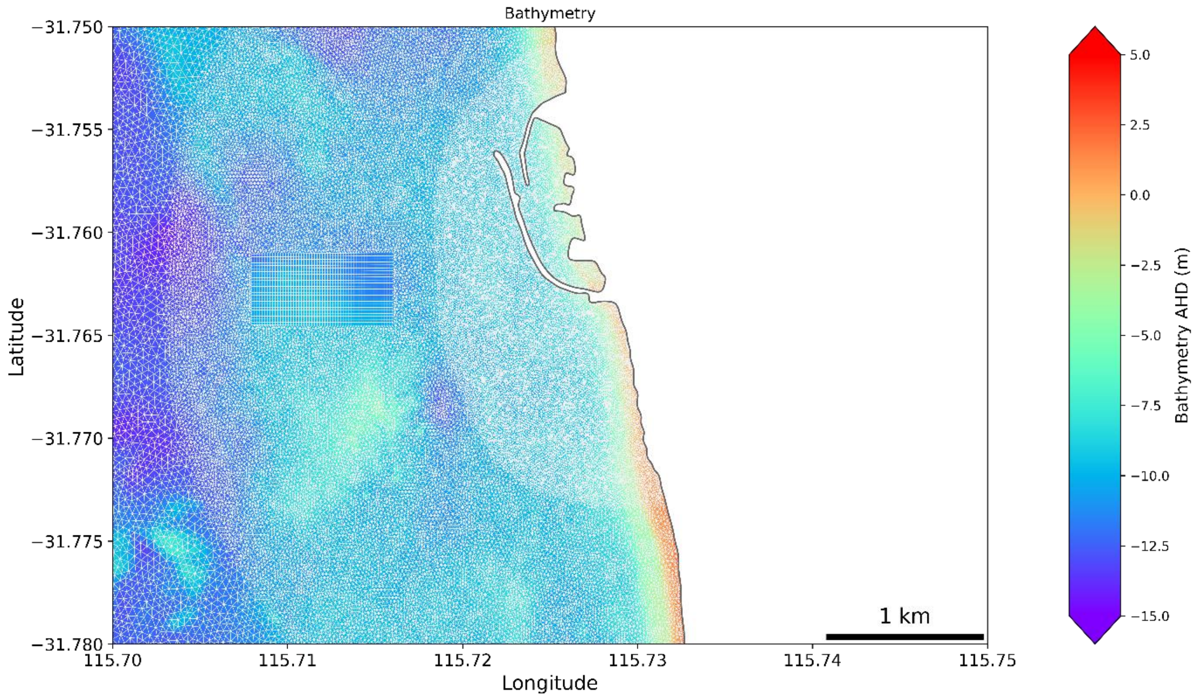


Figure 47: Local Numerical Mesh for Pre- (left) and Post-Marina Development (Right)

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6. Far-Field Model Validation

6.1. Framework for Assessing Model Accuracy

The model accuracy was assessed by means of:

- Qualitative comparison of modelled timeseries against measured parameters.
- Quantitative evaluation of model performance through a set of skill scores.
- Qualitative assessment of progressive vector diagrams.

The set of skill scores considered are:

- model bias:

$$Bias = \frac{1}{n}\sum_{i=1}^{n}[M_i - O_i]$$

- mean squared error:

$$MSE = \frac{1}{n}\sum_{i=1}^{n}[M_i - O_i]^2$$

- root mean squared error:

$$RMSE = \left[\frac{1}{n}\sum_{i=1}^{n}[M_i - O_i]^2\right]^{\frac{1}{2}}$$

- the Wilmott (1981) Index of Agreement:

$$IOA = 1 - \frac{\sum_{i=1}^{n}[M_i - O_i]^2}{\sum_{i=1}^{n}\left[[M_i - \bar{O}] + [O_i - \bar{O}]\right]^2}$$

In these formulae 'n' is the number of observations, M and O represent modelled and observed values, respectively, the subscript i indicates the i^{th} point in the record, and the overbar denotes the arithmetic mean. As each of these equations require that Mi and Oi are contemporaneous, the observed data are transformed by means of linear interpolation in time.

6.2. Validation Scenarios

Model validation was performed for two scenarios representative of the typical seasonal wind conditions in summer and winter (Table 20). Validation periods aligned with available current data at Ocean Reef AWAC sites, which sampled pre-and-post redevelopment of the marina hydrodynamic conditions.

Validation was undertaken for Mesh A.

Table 20: Validation Scenarios

Scenario No.	Modelled Period	Season	Coastal layout	Validation Site
V1	01/01/2019 - 01/04/2019	summer	Pre-development	AWAC1
V2	01/05/2022 - 01/08/2022	winter	Post-development	AWAC2 & SIG_HIL

6.3. Validation Results

6.3.1. Ocean Reef

Time series of modelled water levels, current speed, and current direction were qualitatively compared with 1-month of AWAC1 and AWAC 2 data, as shown in Figure 48 and Figure 49, respectively. Overall, the model performed well during both the summer (V1) and winter (V2) hydrodynamic simulations.

The model accurately captures both the timing (phase) and magnitudes of the tidal components.

Modelled pre-development conditions (simulation V1, Figure 48) compared well with measured data and achieved IOAs of 0.939 and 0.866 for water levels and depth averaged currents, respectively. The RMSE of 0.03 m/s depth averaged currents is deemed low and comparable to the instrument noise floor.

The agreement between modelled (simulation V2, Figure 49) and measured quantities post development during winter was similar to that described for the summer, pre-development simulation. In this case, an IOA = 0.754 for currents and 0.963 for water level was achieved, both values above industry standard acceptable levels. The model reproduces storms well and, generally, events that lead to depth averaged currents >0.05 m/s. Winter, however, presents extended calm to low-wind periods with low wind-driven currents. During such periods, some discrepancies between model and measured data are noted, but these can be explained by the instrument approaching its noise floor rather than poor model performance. The discrepancy may also be attributed to secondary processes, such as long period water level variations (e.g. coastally trapped waves) and wave induced currents, etc., which were not considered in this model as part of the stagnation/mixing study.

Current directions are, overall, well represented by the model which reproduces the dominant seasonal behaviour for summer and winter. The relatively small directional spread in field observations particularly during the V1 modelling period (Figure 49) might once again be explained by the uncertainty surrounding the measurement of current direction at low current speeds.

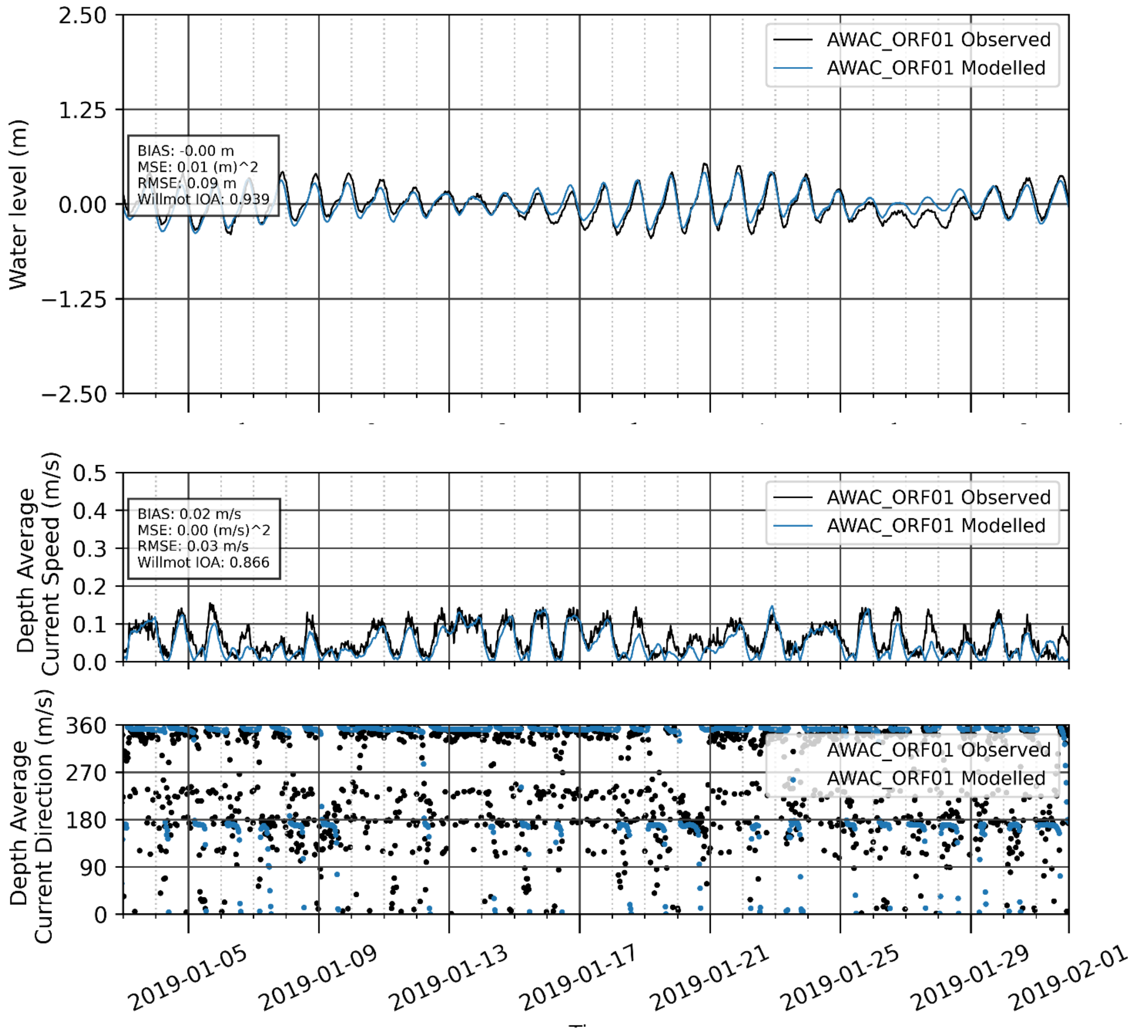

Figure 48: Validation of Water levels, Current Speed and Direction at AWAC_ORF01 during January 2019 (Pre-development)

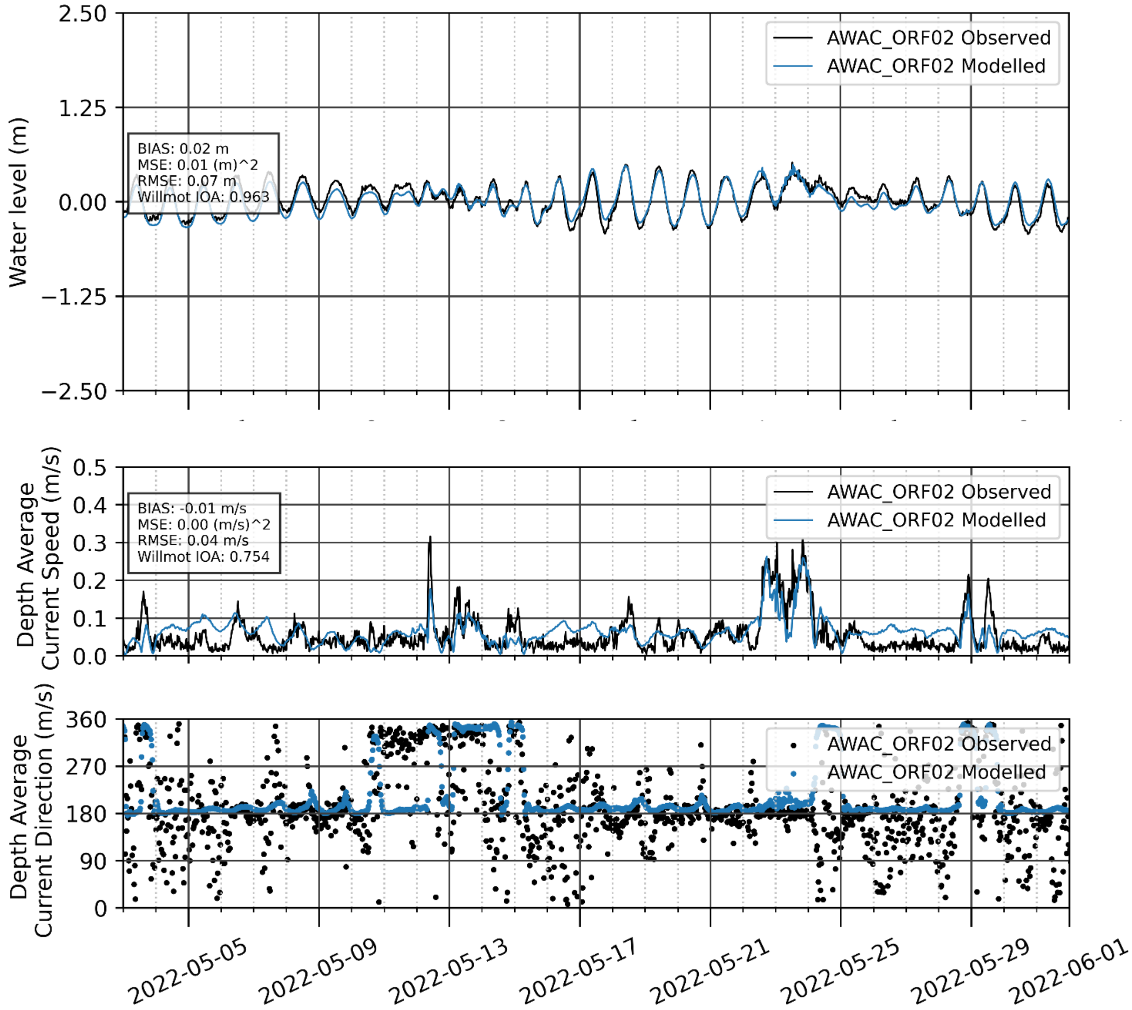

Figure 49: Validation of Water levels, Current Speed and Direction at AWAC_ORF02 during May 2022 (Post-development)

6.3.2. Hillarys

Data from a Nortek Signature deployed offshore Hillarys was available for a 1-month period during the winter covered by simulation V2. Model agreement to measured data at this location was weaker compared to Ocean Reef, with an IOA of 0.971 for water levels and 0.648 for depth averaged current speed. Limited confidence was placed in measured current directions at low speeds, so the mismatch in current directions and magnitude during low-current events shown in Figure 50 should be interpreted with caution. Peak currents were nonetheless well replicated by the wind-driven model.

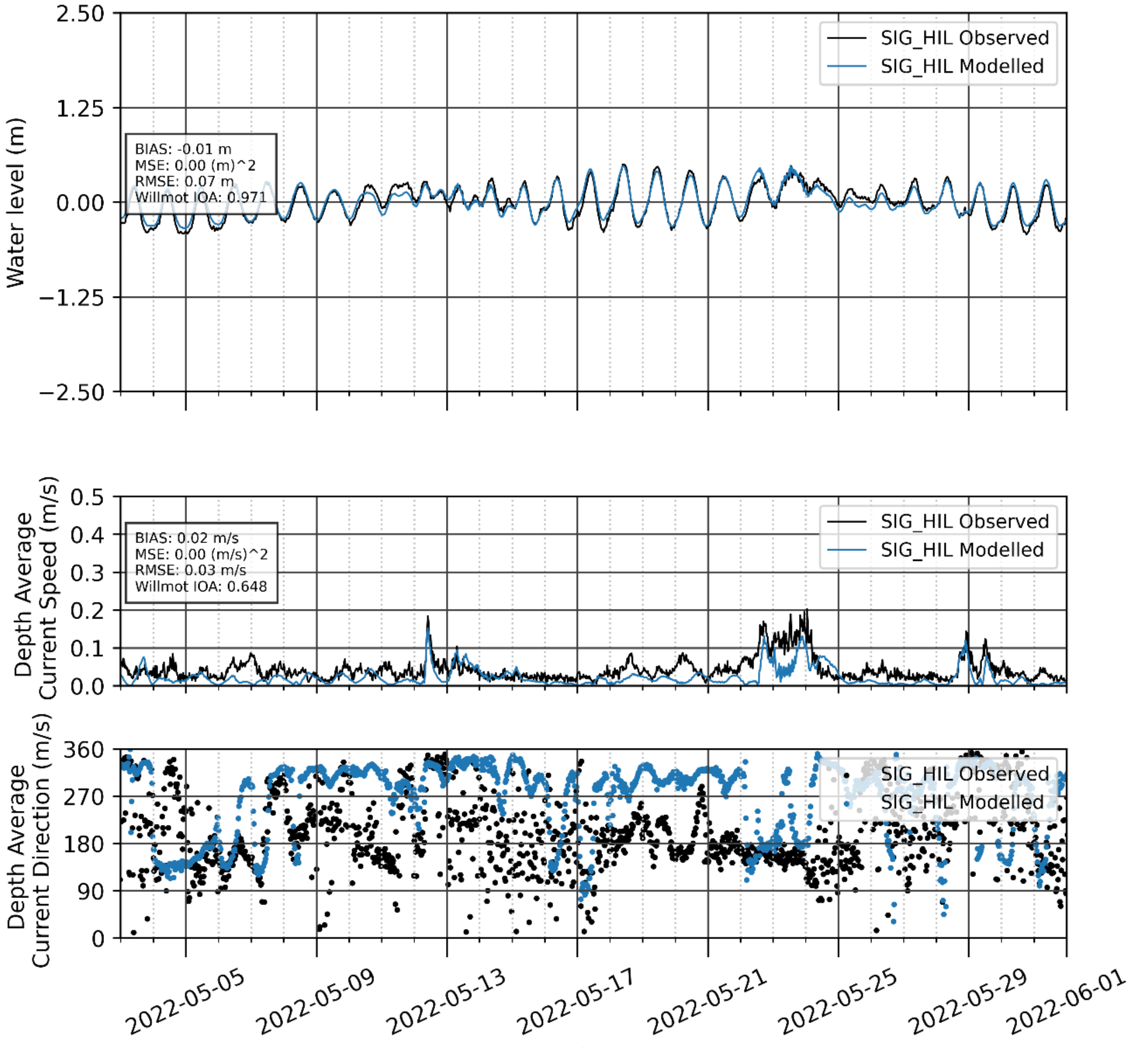

Figure 50: Validation of Water levels, Current Speed and Direction at SIG_HIL during May 2022 (Post-development)

6.3.3. Localised Drift

The model's ability to accurately simulate the low frequency drift currents is essential for this project, as it directly impacts the prediction of the fate and transport of the Beenyup effluent near the marina and surrounding beaches. Resolution of these currents rely on accurate wind-fields and good hydrodynamic model parameterisation.

Progressive vector diagrams (PVD) provide a good qualitative representation of the model's performance in simulating drift currents. It is noted, however, that the model was designed to capture fate and transport processes over length scales of ~2 km and periods of up to ~2 days, the distance which may be influenced by the redevelopment of the Ocean Reef Marina and the time it would take a substance discharged through the Beenyup outfall to cover this extent. Conservatively, PVD plots were generated for 5-day periods aligning with the typical conditions that this study wishes to replicate (summer and winter seasonal patterns), within simulations V1 and V2.

Summer conditions are relevant to this investigation as it is the most likely period to develop algal blooms and other water quality issues. Model performance over a 5-day summer period is shown in Figure 51. A sustained north-easterly drift current results in a PVD hypothetical distance of ~27 km. The spread between model and measured hypothetical end points over this period is ~1.7 km, or 6% of the total distance. When a 24-hr period is considered, the spread is equal to ~270 m, also 6% difference between model and measured data. The divergence between the modelled and measured PVD appears consistent on a day-to-day basis, owing to the cyclical daily summer flow patter, which is well represented by the model. Consequently, the model's representation of the fate and transport of the Beenyup discharge under typical summer conditions is estimated to achieve an accuracy of approximately 120 m over a 2 km distance from the discharge point.

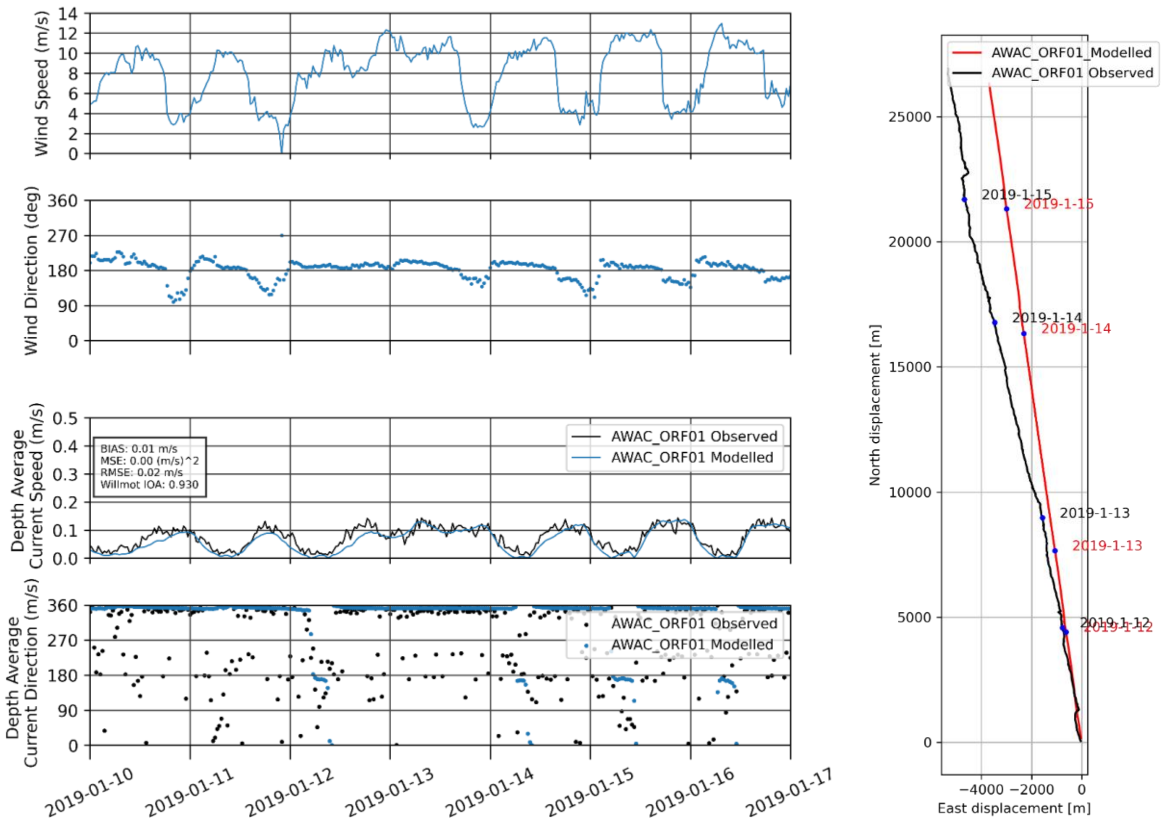

Figure 51: Forcing wind conditions (top left), modelled and measured current speed and direction (bottom left), and Progressive Vector Diagram (PVD), right during a typical summer net drift condition (Simulation V1, pre-development)

7. Far-field Model Application

7.1. Scenarios

A total of 10 scenarios (Table 21) were designed to target the community concerns highlighted in Section 1.2. The list of scenarios were agreed by DWER.

The scenarios focus on:

1. The effect of the marina on local hydrodynamics (Section 7.2)
2. The effect of the marina on the fate and transport of the Beenyup effluent (Section 7.3)

Table 21: Simulation Scenarios run for Mesh A

Scenario	Modelled Period (UTC)	Marina Status	Wind Forcing Description	Beenyup Discharge Flow Rate and Behaviour	Rationale
V1	01/01/2019 to 01/04/2019	Pre-Development	BoM Ocean Reef Observations	Not represented	Summer validation of the model and pre-development baseline conditions for comparison to S1.
V2	01/05/2022 to 01/08/2022	Post-Development	BoM Ocean Reef Observations	Not represented	Winter validation of the model and pre-development baseline conditions for comparison to S2.
S1	01/01/2019 to 01/04/2019	Post-Development	BoM Ocean Reef Observations	Not represented	Estimate of local summer hydrodynamic conditions post marina development, for comparison with V1.
S2	01/05/2022 to 01/08/2022	Pre-Development	BoM Ocean Reef Observations	Not represented	Estimate of local winter hydrodynamic conditions pre-marina development, for comparison with V2.
S3	01/01/2019 10:00 to 03/01/2019 22:00	Pre-Development	Idealised conditions of constant and uniform winds: • Direction = 315 ° (north-west) • Speed = 4.28 m/s (Table 2)	Summer daily discharge pattern (Figure 34), split evenly among diffuser nozzles at seabed. Discharge starts at 01/01/2019 10:00 UTC (06:00 AM local) following model soft start period.	Assessment of the fate and transport of Beenyup effluent under constant (persistent) north-westerly wind pattern and summer temperature and salinity conditions. Comparison between pre- and most marina development to investigate
S4		Post-Development			

					any potential influence of the new marina on effluent dispersion.
S5	01/01/2019 10:00 to 03/01/2019 22:00	Pre-Development	Idealised conditions of constant and uniform winds: • Direction = 270° (west) • Speed = 5.47 m/s (Table 2)	Summer daily discharge pattern (Figure 34), split evenly among diffuser nozzles at seabed. Discharge starts at 01/01/2019 10:00 UTC (06:00 AM local) following model soft start period.	Assessment of the fate and transport of Beenyup effluent under constant (persistent) westerly wind pattern and summer temperature and salinity conditions. Comparison between pre- and most marina development to investigate any potential influence of the new marina on effluent dispersion.
S6		Post-Development			
S7	01/01/2019 10:00 to 03/01/2019 22:00	Pre-Development	Idealised conditions of constant and uniform winds: • Direction = 225 ° (south-west) • Speed = 7.19 m/s (Table 2)	Summer daily discharge pattern (Figure 34), split evenly among diffuser nozzles at seabed. Discharge starts at 01/01/2019 10:00 UTC (06:00 AM local) following model soft start period.	Assessment of the fate and transport of Beenyup effluent under constant (persistent) south-westerly wind pattern and summer temperature and salinity conditions. Comparison between pre- and most marina development to investigate any potential influence of the new marina on effluent dispersion.
S8		Post-Development			

7.2. Effect of the Marina on Local Hydrodynamics

The effect that the post-development Ocean Reef Marina has on local hydrodynamics was investigated by computing the difference in modelled current speeds relative to pre-development conditions during representative summer and winter regimes. The difference in current speeds were calculated for each numerical grid and time step, for the following pair of Scenarios:

- Summer conditions: pre-development Scenario V1 and post-development Scenario S1.
- Winter conditions: pre-development Scenario V2 and post-development Scenario S2.

Hourly 'difference' plots over the full simulation period were then generated.

7.2.1. Summer Conditions

Although hundreds of hourly difference plots were generated, O2Me selected a representative 24-hr period to illustrate hydrodynamic changes caused by the post-development marina compared to pre-development conditions. To capture typical hydrodynamic variations under summer wind patterns, depth averaged current speeds modelled for 17th of January 2019 were analysed. The difference in currents is shown in Figure 52. During the mornings, wind-driven circulation has barely changed post-development of the marina owing to the limited fetch disruption caused by the marina westward of the coastline. Later in the day, the wind rotates and the interaction of the new marina layout with south-westerly seabreeze leads to a 0.05-0.10 m/s net reduction of current speeds leeward of the marina (blue-toned patch north of the development, Figure 52). A relatively small zone south of the marina is also affected and currents are lower than during pre-development conditions. Alongshore northerly flows must negotiate the new marina resulting in a modest increase in current velocities west of the redevelopment, extending toward the Beenyup diffuser site (red-toned patch west of the marina, Figure 52).

Quantification of the relative reduction of depth averaged currents around the Ocean Reef Marina and nearby beaches post-redevelopment, relative to pre-redevelopment conditions, is shown in Figure 53. The figure was prepared for a typical summer day (17/01/2019 14:00). The figure revels that:

- Reductions in depth averaged currents are larger to the north of the marina than they are to the south.
- The 10% ([8]) reduction contour extends approximately 1500 m to the north and 500 m to the south of the marina. The actual extent of this contour varies depending on the time of day and prevailing wind conditions, so it should be interpreted as a relative value rather than an absolute one.
- The length scale associated with the 40% reduction in depth averaged currents on the selected day representative of summer conditions is substantially shorter than that of the 10% contour, with approximate lengths of 700 m to the north and 50 m to the south of the marina.

[8] This and other similar percentages adopted to describe changes to the current regime in this section were arbitrarily selected for discussion.

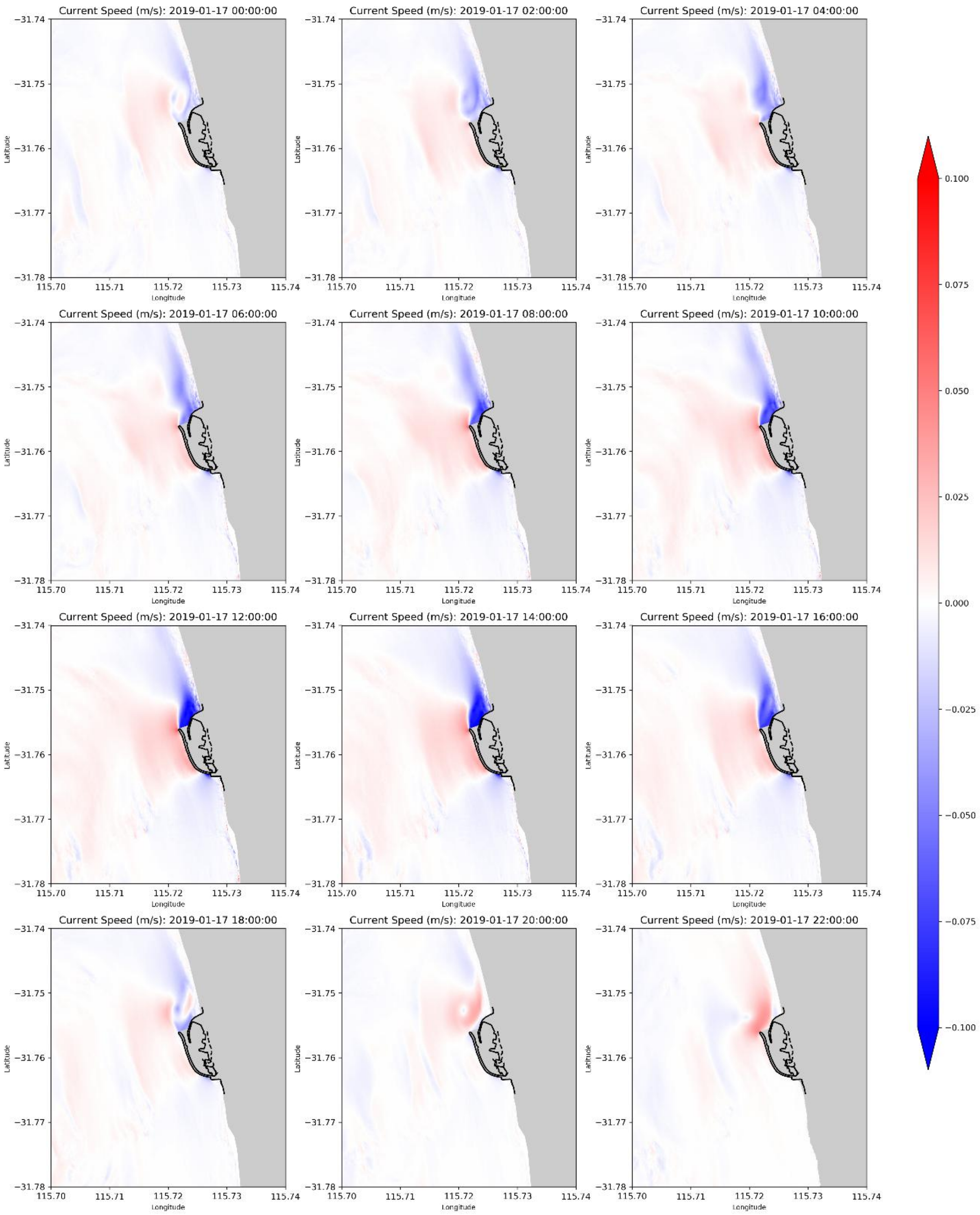

Figure 52: Difference in depth averaged current speed associated with the Marina Redevelopment (Post-development minus Pre-Development) for a typical summer day

Figure 53: Reduction of depth averaged currents post-redevelopment relative to pre-redevelopment conditions, during a typical summer day (17/01/2019 14:00).

7.2.2. Winter Conditions

Two typical winter patterns were considered: extended periods of calm wind conditions and short storms. This section examines the influence of the redeveloped marina on local hydrodynamics under both conditions. June 8th, 2022, was selected as representative of the relatively calm period (Figure 54), and May 23rd, 2022, was chosen to represent the hydrodynamics of short-lived storms (Figure 55).

Current flows around the redeveloped marina during a typical calm day in winter shares resemblance with the pattern experienced during a summer day: depth averaged currents are lower to the north and south of the marina relative to pre-development conditions, while increased currents are evident to the west of the marina. Unlike the summer scenario, however, the following can be said:

- The pattern remains relatively consistent throughout the day, which is expected given that the wind forcing is often persistent from a northerly direction.
- The marina causes disruption of a southerly current flow during winter, and hence a slightly larger area of depth average current reduction is noted south of the marina, compared to the reduction north of the marina.
- The net differences in depth averaged current speed reduction during a typical winter day are smaller than the reductions observed during a typical summer day.

During storm events, a similar behaviour to that observed on a typical winter day is seen, although the difference in current magnitudes is more pronounced (Figure 55). While the net difference is much larger during storms, these events are infrequent and short-lived. Water quality issues are unlikely to be a concern during these events due to extensive mixing. It is noted that public concerns are rarely raised on these days.

Quantification of the relative reduction of depth averaged currents around the Ocean Reef Marina and nearby beaches post-redevelopment, during the typical (relatively calm) winter day, is shown in Figure 56, prepared for a typical summer day (08/06/2022 14:00). The following observations are noted:

- Although absolute current magnitude reduction is much smaller in winter than in summer, the relative reduction in current magnitude (percentage difference) is generally larger than those observed during summer, owing to generally lower current speeds.
- Reductions in current magnitude of 10% extend approximately 1,100 m and 1,200 m north and south of the marina, respectively.
- The length scale associated with a 40% reduction in magnitude decreases substantially to 300 m and 500 m, for the north and south of the marina respectively.

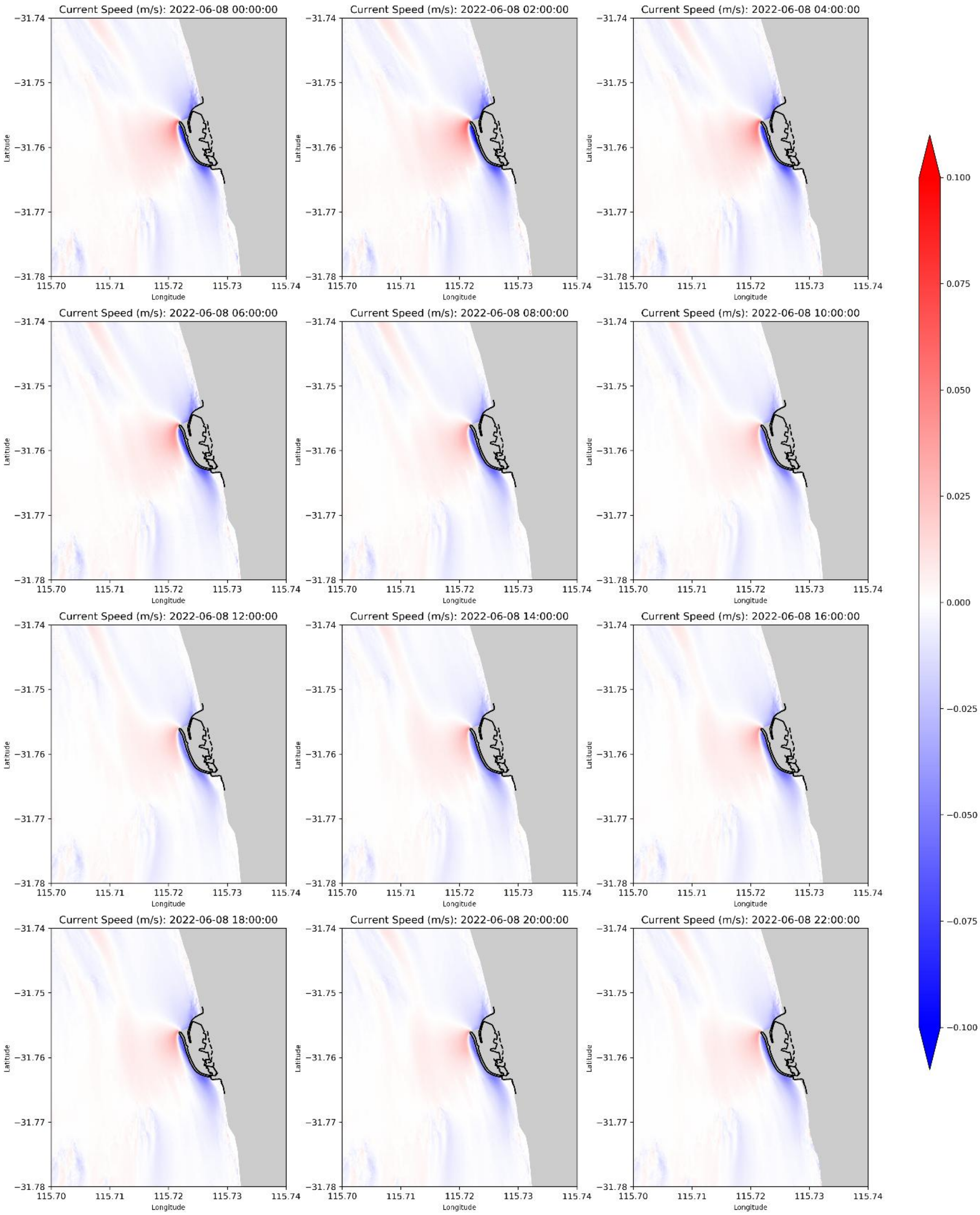

Figure 54: Difference in depth averaged current speed associated with the Marina Redevelopment (Post-development minus Pre-Development) for a typical winter day: relative calm conditions

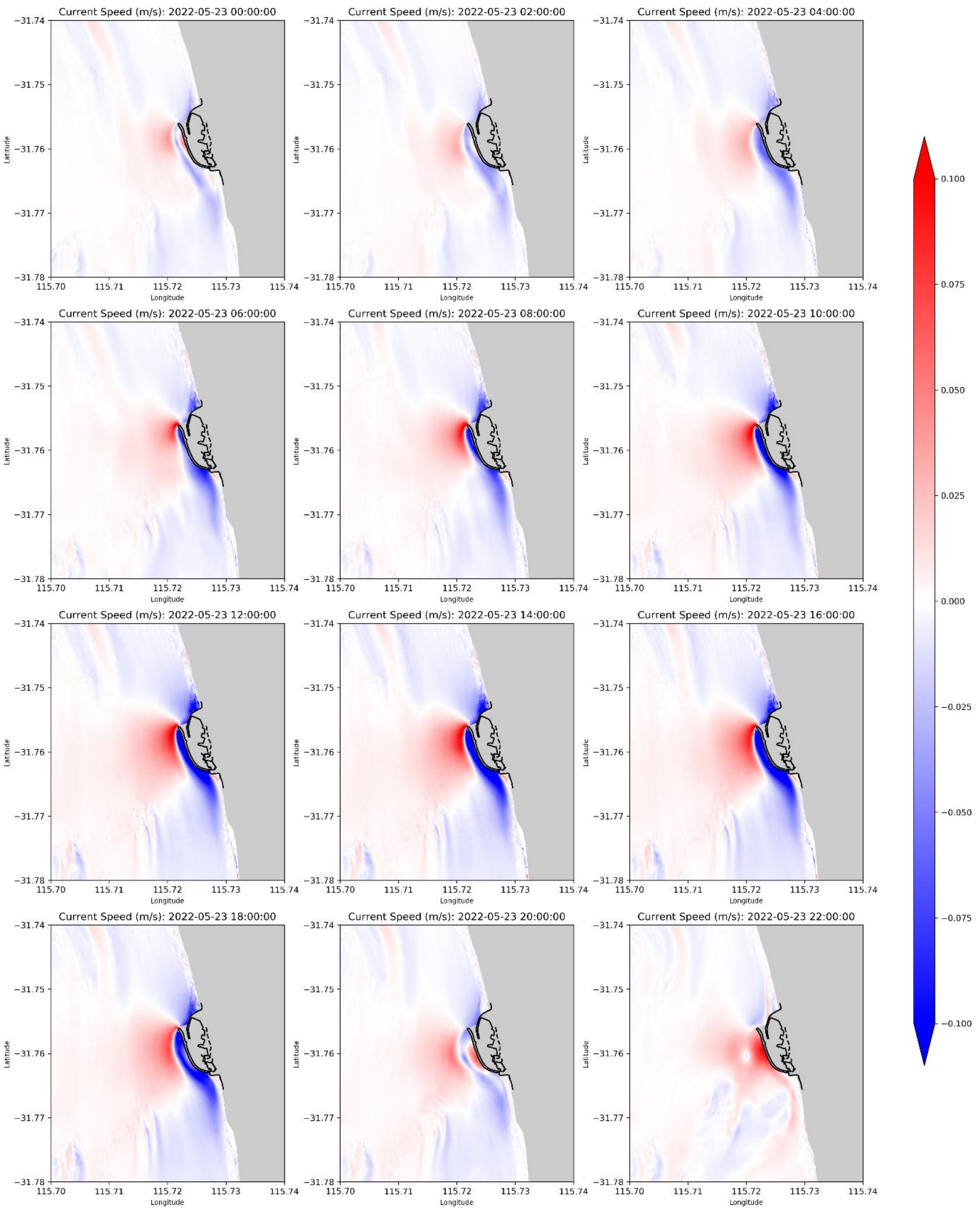

Figure 55: Difference in depth averaged current speed associated with the Marina Redevelopment (Post-development minus Pre-Development) for a typical winter day: short-lived storm

Figure 56: Reduction of depth averaged currents post-redevelopment relative to pre-redevelopment conditions, during a typical winter day, calm winds (08/06/2022 14:00)

7.3. Effect of the Marina on Fate and Transport of Beenyup Discharge

A qualitative assessment of the fate and transport of the Beenyup effluent is presented in this section. The assessment examined the predominant direction and spatial extent at the surface and near seabed layers, of an idealised effluent released through the Beenyup diffusers. Contours representing the 95% non-exceedance thresholds for various dilution factors (10/50/100/200/300/400/500 fold) and time since discharge (12, 24, and 36 hours) were mapped as part of the assessment. An O2Me analyst qualitatively compared the results of the pre-redevelopment and post-redevelopment configurations to evaluate whether the redevelopment of the Ocean Reef Marina may have affected the mixing of the Beenyup effluent in seawater.

Modelling scenarios were forced with idealized wind conditions (Table 21). The decision to avoid replicating specific events was attributed to the following reasons:

- Impracticality of capturing all possible scenarios: Attempting to replicate every possible wind condition or water quality event of public concern would be unfeasible due to the sheer number of variables and the dynamic nature of the coastal systems.
- Focus on patterns that would align with public concerns: The modelling targets conditions that are most likely to cause interaction between the Beenyup discharge and the regions where public concern has been noted during summer periods (Figure 5). This allows enables the analysis for both frequent and infrequent summer conditions in isolation, helping to understand how long these conditions must be sustained for significant interactions to occur. Once identified, the feasibility of sustaining the required duration can be assessed and discussed.
- Avoiding event-specific bias: Modelling specific events might skew the results towards scenarios that may not reoccur, or that can potentially misrepresent the broader effects of the coastal environment in question.
- Generality and transferability of results: Trends derived from idealized conditions provide insights that are more widely applicable across different temporal and spatial contexts, enhancing the relevance of the findings.
- Simplification of complex interactions: Idealized conditions allow the study to isolate key processes influencing effluent fate and transport, avoiding confounding factors introduced by highly variable, complex factors.

7.3.1. Results

The following observations are made:

- The Beenyup effluent spreads over a larger area at the water surface compared to near the seabed.
- Under all wind patterns and durations considered, there are negligible differences in the extent and location of the dilution contours when post-redevelopment results are compared to pre-redevelopment conditions (Figure 57 to Figure 62).
- Persistent **north-westerly** winds drive the Beenyup effluent south, parallel to Mullaloo and Ocean Reef Beaches (Figure 57 and Figure 58).
- Persistent **south-westerly** winds drive the Beenyup effluent north. Over typical south-westerly seabreeze periods of <12 hr and up to a-typical periods of up to ~24 hr, effluent waters remain far from the coastline (left and centre panels in Figure 60). The effluent would be trapped in the

eddy at the head of the marina, and eventually reach Burns Beach area, if the wind persisted for an a-typical period of ~36 hr (right panels in Figure 60).

- Persistent **westerly** winds drive the Beenyup effluent south-east, towards Ocean Reef and Whitfords Beach (Figure 61 and Figure 62). For the effluent to reach the coastline, an a-typical persistent summer westerly wind lasting >36 hr would be required.

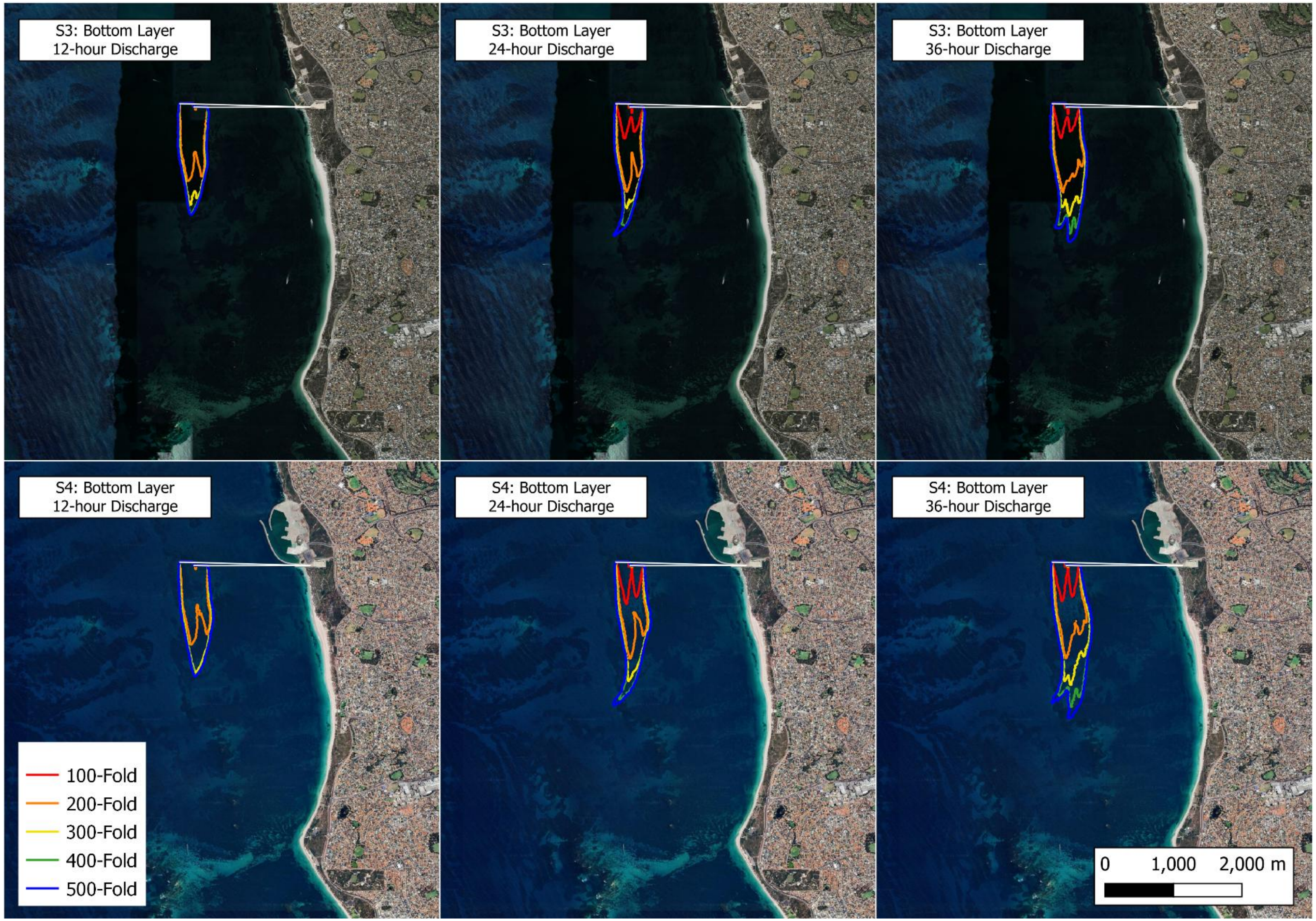

Figure 57: 95th Percentile seabed dilution contours for 12, 24 and 36 hours of constant NW forcing Pre-Development (S3) and Post Development (S4)

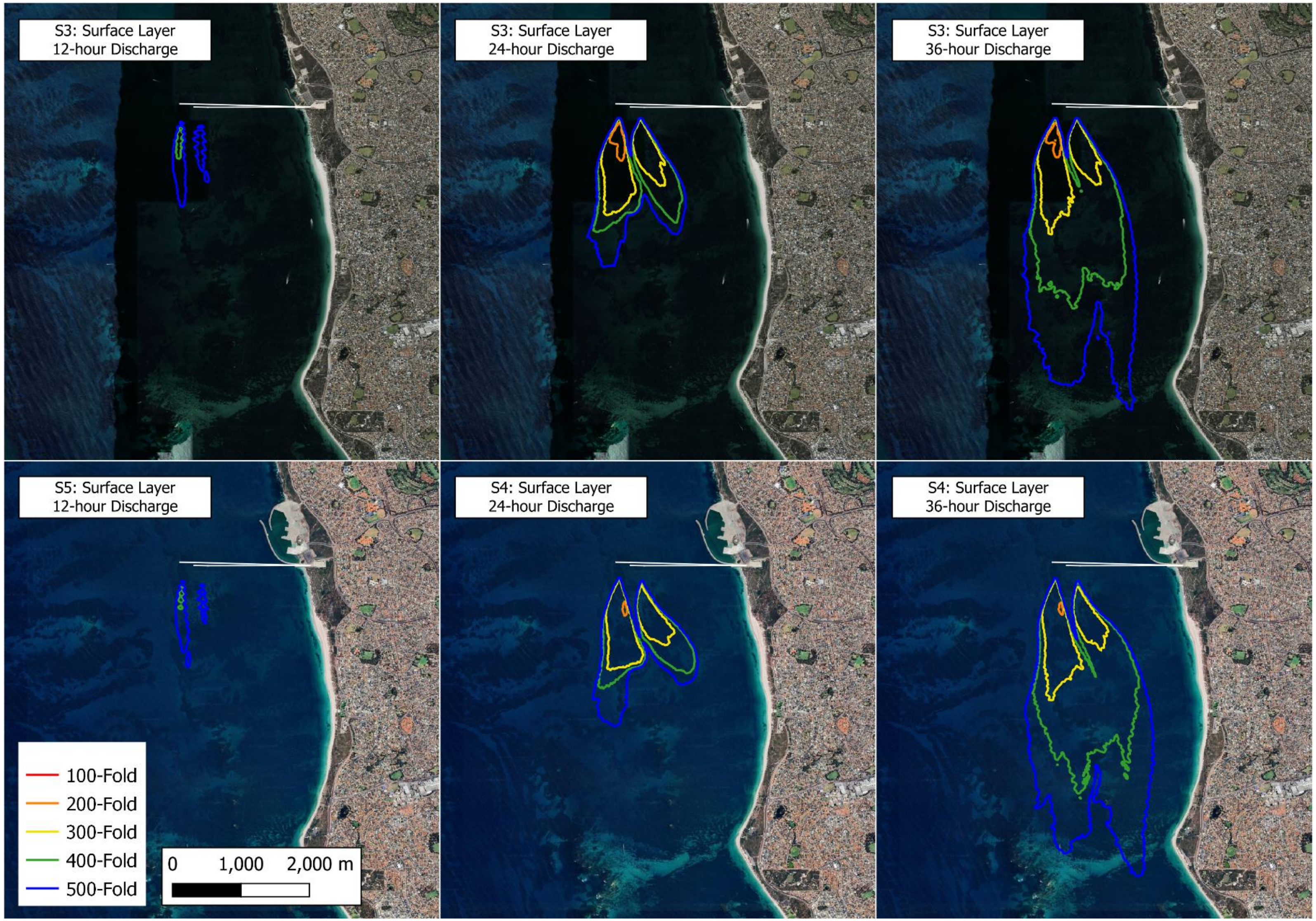

Figure 58: 95th Percentile surface dilution contours for 12, 24 and 36 hours of constant NW forcing Pre-Development (S3) and Post Development (S4)

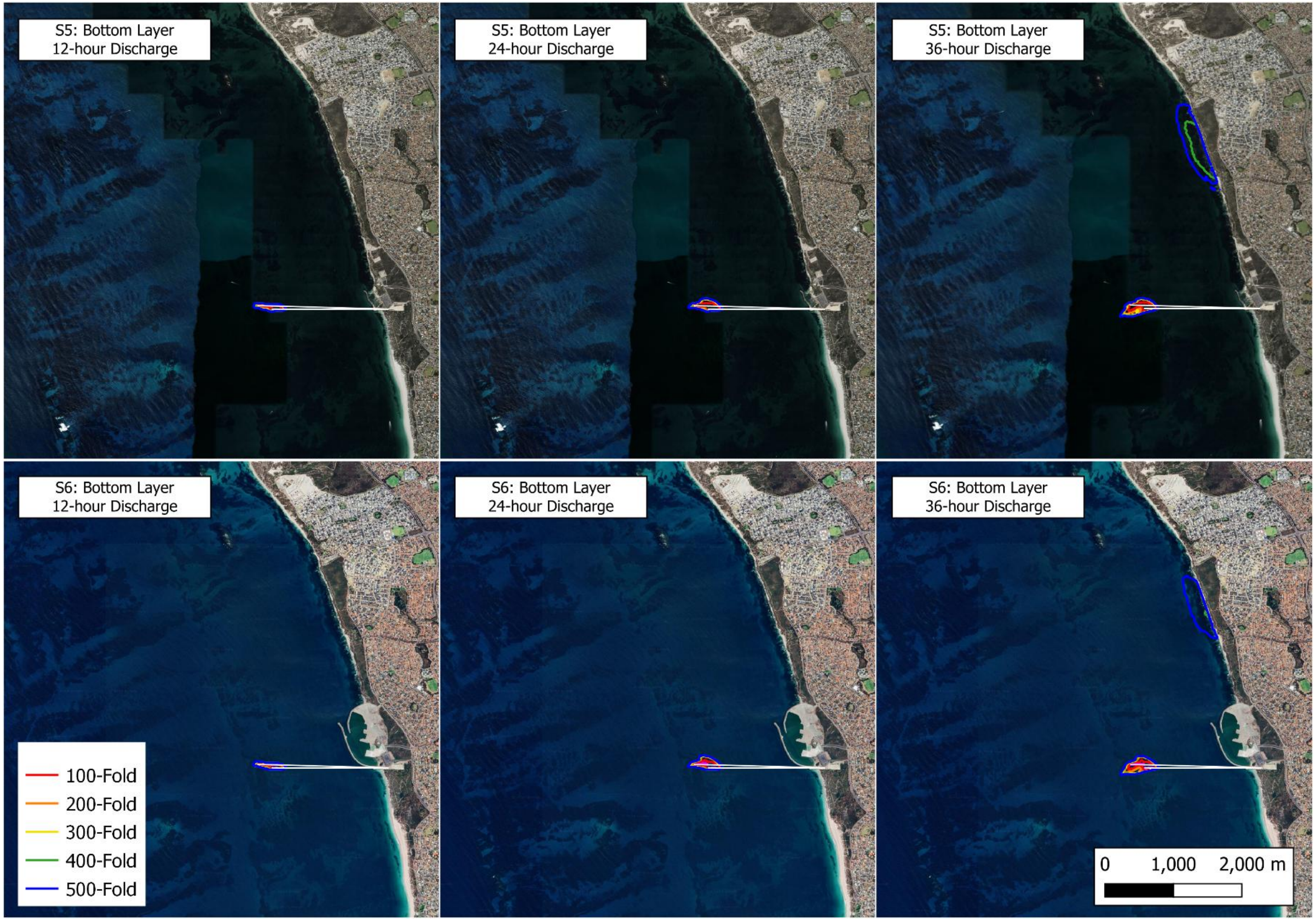

Figure 59: 95th Percentile seabed dilution contours for 12, 24 and 36 hours of constant SW forcing Pre-Development (S5) and Post Development (S6)

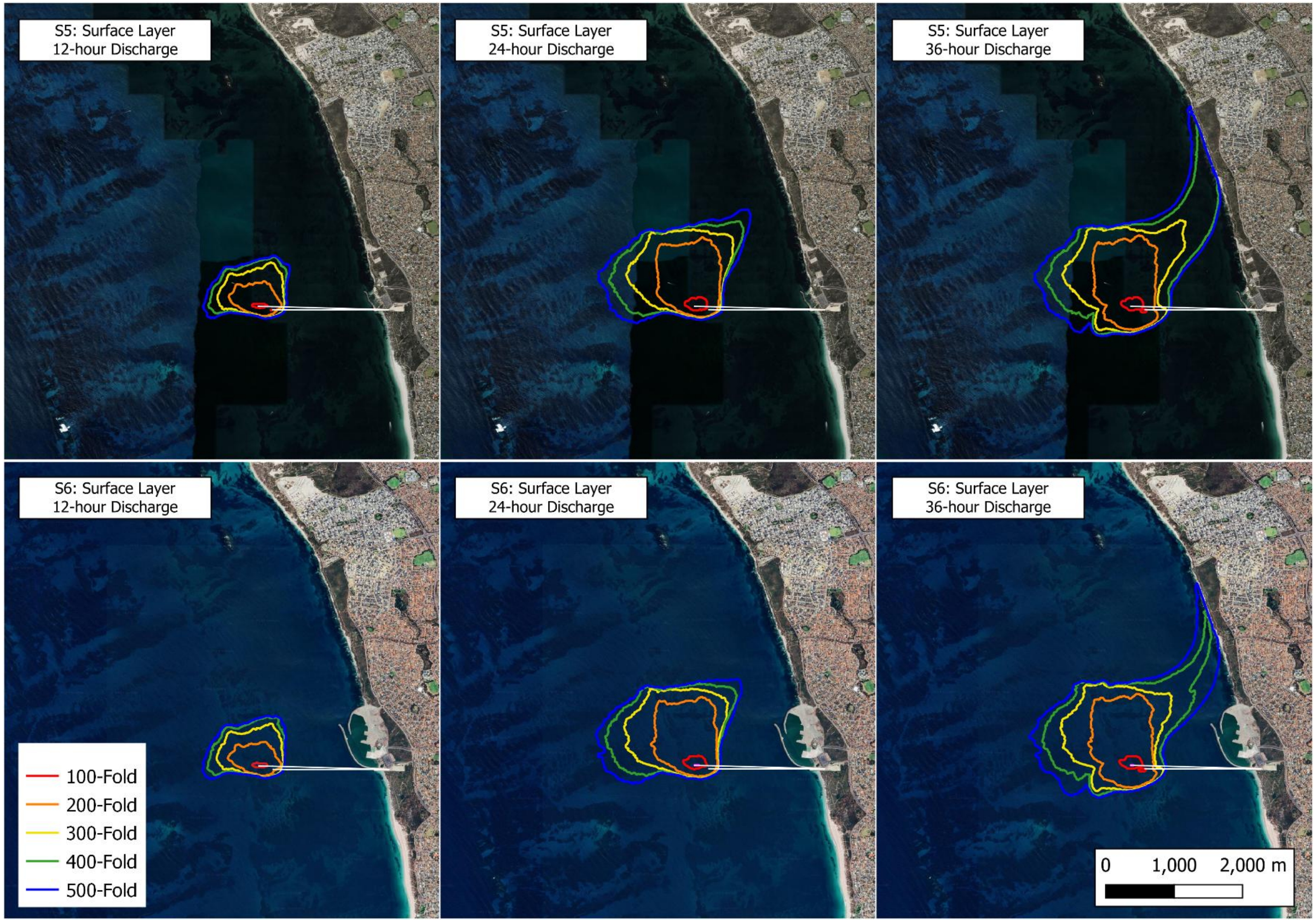

Figure 60: 95th Percentile surface dilution contours for 12, 24 and 36 hours of constant SW forcing Pre-Development (S5) and Post Development (S6)

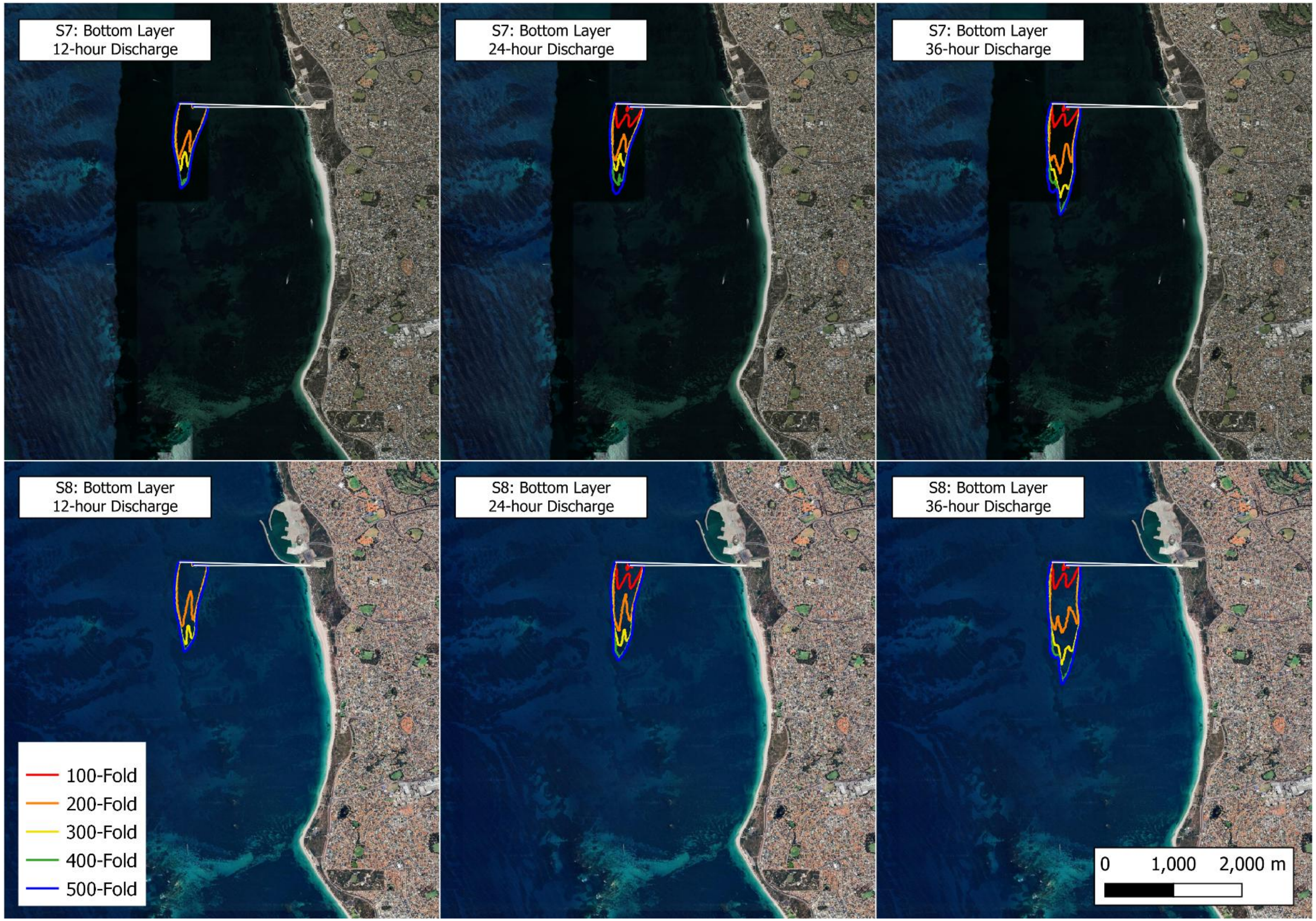

Figure 61: 95th Percentile seabed dilution contours for 12, 24 and 36 hours of constant W forcing Pre-Development (S7) and Post Development (S8)

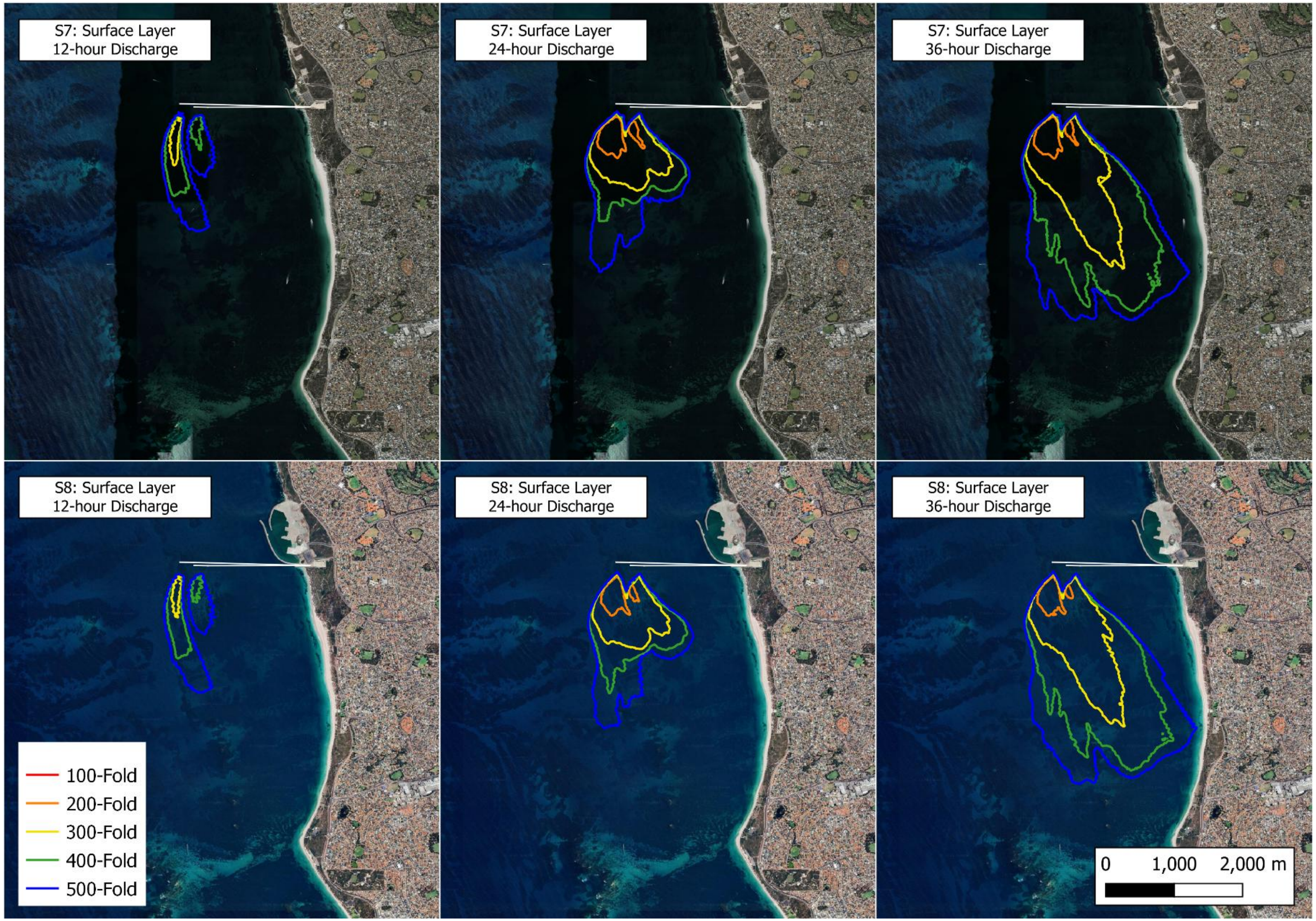

Figure 62: 95th Percentile seabed dilution contours for 12, 24 and 36 hours of constant W forcing Pre-Development (S7) and Post Development (S8)

8. Conclusion

A variety of water quality datasets were reviewed to characterise changes in the composition of the TWW discharge and assess its influence on marine water quality in relation to recent issues reported at Mullaloo beach and surrounding beaches. A clear and consistent explanation of Beenyup ocean outfall contribution to recent marine water quality issues cannot be provided, due to limitations in the available data for evaluation

The 'WA Health watch list' Cyanobacteria *Trichodesmium* was present in samples from Mullaloo Beach on 17 January, and 24 and 26 of April 2024, which is likely the cause of the brown surface scum identified in public complaints. An algal bloom of *Trichodesmium* is consistent with water quality observations received from the public, including appearance of the slick, colour, toxic health risk and odour. *Trichodesmium* blooms are common in coastal Australia. These algae fix airborne nitrogen and convert into bioavailable compounds for phytoplankton in the water column. Unlike other microalgae, *Trichodesmium* is not necessarily produced from an increase in nutrient loads in water. Therefore, the sites of concern may present naturally conducive conditions for this species, noting that a *Trichodesmium* bloom had already been reported two years prior. A practical solution to recurring algal blooms may not be feasible if these events are naturally produced.

Trichodesmium blooms are seasonal and require suitable water temperatures, sources of iron (Fe) to grow and fix nitrogen, black carbon aerosols from frequent bushfires (Qi et al, 2023), and phosphorus for growth and fixation (Chinenye, 2023). Elevated nitrogen and phosphorus in TWW gradually reduce with distance from the outfall in the direction of the prevailing currents and are barely detectable at concentrations above background 1,500 m downstream. These plumes rarely track towards Mullaloo Beach in summer. Therefore, nutrient inputs to the beach south of the redeveloped marina from the Beenyup discharge are likely to be minimal.

Natural background concentrations for chlorophyll-a in the region have increased over the last three years, which raised nutrient loading concerns. However, discharge operations have not substantially changed and regular monitoring has not detected substantial discharge elevations in nutrients. In addition, elevated *Enterococci* concentrations recorded at shoreline sites on 23 January 2024 suggest a source of faecal matter present isolated from the Beenyup outfall.

Water quality is influenced by several natural and anthropogenic factors, which may include changes in local winds, current speeds, local groundwater seepage and nonpoint source pollution from the marina. Further assessment, for example targeted sampling, would be required to help understand the complexity of the system beyond the review undertaken in this study.

The introduction of the revised ocean reef marina reduces current velocities north and south of the marina during typical summer and winter conditions. In this study, the zone of influence of the marina was arbitrarily described using the contour of 10% reduction in depth averaged currents modelled for representative summer and winter conditions. Model simulations suggest that this contour extends approximately 1,500 m to the north and 500 m to the south of the marina during a typical summer day. Similarly, the contour extends approximately 1,100 m to the north and 1,200 m to the south of the marina during a typical winter day. These results align with anecdotal reports of areas experiencing poor water quality.

The influence of the redeveloped marina on current flows described in this study should be interpreted as a relative value rather than an absolute one. Whether the modelled reduction in current velocities is substantial enough to create an environment which exacerbates algal bloom events are yet to be confirmed, pending the assessment of other contributing factors, which was out of this scope of work.

The idealised cases modelled provided no evidence of the Beenyup effluent interaction with the coastline north and south of Ocean Reef Marina at up to 500-fold dilutions subject to feasible, persistent, summer wind conditions of up to 12 hr duration. The modelling results agree with the review of historical data which indicates that nutrient concentrations are rarely above the background equilibrium at 1,500 m from the outfall in the direction of the current and, on those occasions, concentrations are unlikely to represent measurable effects on water quality. In addition, the numerical results of idealised conditions suggest that atypical summer metocean circumstances would be required for treated wastewater discharged through the Beenyup outfall to reach the coastline at concentrations below 500-fold dilution. For example, wind events persisting for more than 24 hours (occurs approximately 1% of the time during summer, or approximately once over this period) from the north-west direction (experienced less than 3% of the time during summer) would be required.

While the continuous nutrient loading from TWW discharge since the late 1970's may have contributed to the background nutrient equilibrium, dispersion modelling of a numerical signal representative of the idealised Beenyup effluent revealed that under the wind directions and intensities considered, there were negligible differences in the extent and location of the Beenyup effluent dilution contours when post-redevelopment results were compared to pre-redevelopment conditions. No evidence has been found to support the claim that the marina redevelopment has substantially altered the distribution of the dispersed Beenyup effluent near the coastline, and therefore it is unlikely solely responsible for the alleged decline in marine water quality or high concentrations of *Trichodesmium* algae.

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Appendix A. Public submissions by geographical location during the submission period (August 29 – September 30, 2024)

Geographic reference	Number of submissions
Mullaloo (beach/shoreline)	55
Mullaloo SLSC	3
Key West car park (north Mullaloo Beach)	1
Pinnaroo Point	2
Ocean Reef (beach, boat harbour, marina)	13
Iluka (beach)	2
Burns Beach	3
Whitfords beach	2
Whitfords lagoon	5
Lal bar	1
Little Island	1
Hillary's marina	2
Sorrento beach	1
Quinns Rocks	1
Marmion Marine Park	1
No geographic information	11
Total number of submissions	73